# Title Page

**Plant community compositional stability over 40 years in a Fraser River Estuary tidal freshwater marsh**

Stefanie L. Lane1, Nancy Shackelford2, Gary E. Bradfield3, Madlen Denoth3, 4, Tara G. Martin1

1Conservation Decisions Lab, Department of Forest and Conservation Science, University of British Columbia, Vancouver, BC, Canada; 2School of Environmental Studies, University of Victoria, Victoria, BC, Canada; 3Department of Botany, University of British Columbia, Vancouver, BC, Canada; 4Gymnasium Neufeld, Bern, Switzerland

Corresponding author: [stefanielane@utexas.edu](mailto:stefanielane@utexas.edu)

ORCID ID

Stefanie L. Lane (corresponding author): [0000-0002-4851-2772](https://orcid.org/0000-0002-4851-2772)

Nancy Shackelford: [0000-0003-4817-0423](https://orcid.org/0000-0003-4817-0423)

Tara G. Marin: [0000-0001-7165-9812](https://orcid.org/0000-0001-7165-9812)

## Abstract

Long-term data sets documenting temporal changes in vegetation communities are uncommon, yet imperative for understanding trends and triggering potential conservation management interventions. For example, decreasing species diversity and increasing non-native species abundance may be indicative of decreasing community stability. We explored long-term plant community change over a 40-year period through the contribution of data collected in 2019 to two historical datasets collected in 1979 and 1999 to evaluate decadal changes in plant community biodiversity in a tidal freshwater marsh in the Fraser River Estuary in British Columbia, Canada. We found that plant assemblages were characterized by the similar indicator species, but most other indicator species changed, and that overall α-diversity decreased while β-diversity increased. Further, we found evidence for plant assemblage homogenization through the increased abundance of invasive species such as yellow flag iris (*Iris pseudacorus*), and reed canary grass (*Phalaris arundinacea*). These observations may inform concepts of habitat stability in the absence of direct anthropogenic disturbance, and corroborate globally observed trends of native species loss and non-native species encroachment. Our results indicate that within the Fraser River Estuary, active threat management may be necessary in areas of conservation concern in order to prevent further native species biodiversity loss.

## Keywords

shifting baselines; reference conditions; dispersal networks; species turnover; conservation land management

## Acknowledgements

We are grateful to Z. Davis for providing R programming support, to P. Roper for 2019 field assistance, and to B. Staines (Ladner Harbour Authority) provided canoe and harbor access for all field navigation. We thank J. S. Richardson for advising on the 2019 data collection methodologies. We are grateful to M. O’Connor and D. Stewart for providing comprehensive reviews of this manuscript. Research site access was granted by The Ministry of Forests, Lands, Natural Resource Operations and Rural Development.

# Introduction

In a time of rapid global change, temporal shifts in plant community composition can indicate ecosystem stress response and inform conservation management interventions. Shifts in community-dominant species may be indicative of interspecific interactions such as facilitation (Bruno, 2000), succession (Butzeck et al., 2016), or cycles of population dynamics (Holling, 1973). Alternatively, changes in community-dominant species paired with loss of native species diversity and increasing abundance of non-native species may indicate loss of stability through loss of functional redundancy (Donohue et al., 2016; Tilman, 1999; Palmer et al., 1997). In turn, this may indicate reduced resistance to change or capacity to recover from disturbance, known as resilience (Tilman et al., 2006; Bai et al., 2004). Furthermore, the local loss of native species may have stronger negative impacts on regional biodiversity persistence when the regional pool of potential species is reduced or environmentally constrained (Lepš, 2004; Hanski, 1982). Characterization of plant community changes on decadal timescales contributes to observation of meaningful long-term patterns of compositional stability, and is instructive for developing hypotheses to test drivers of disturbance, especially in data-deficient, dynamic landscapes heavily impacted by anthropogenic activities such as estuaries (Underwood et al., 2000; Ovaskainen et al., 2019).

Estuaries are at the terrestrial-marine interface where hydrogeomorphic and ecological changes occur according to daily and monthly cycles, as well changes due to ecosystem scale processes such as sedimentation or marsh subsidence on annual, decadal, and millennial timescales (Pasternack, 2009). Estuarine habitats support high species richness, including species at risk (Kehoe et al., 2021) and are important carbon reservoirs (Gailis et al., 2021; Douglas et al., 2022). Because these ecosystems will experience accelerated change under sea level rise, they are of increasing conservation concern (Brophy et al., 2019); understanding estuarine habitat changes and implications for habitat stability can inform global change resilience strategies. Estuaries in North America are of particular conservation importance in the Pacific Northwest (PNW) because their pathways of retreat or expansion are often spatially restricted by fjord topography (Emmett et al., 2000), whereas estuaries along the Atlantic coast may spread along expansive coastal plains. Tidal freshwater marshes are the upper reaches of estuaries dominated by riverine freshwater, and in the PNW they are particularly important as early transitional habitat along a salinity gradient for anadromous salmonids (Davis et al., 2021; Chalifour et al., 2019). The Fraser River Estuary is the largest estuary in British Columbia and of irreplaceable ecological and commercial value, yet has lost 85% of floodplain and 64% of stream habitat in the Lower Fraser watershed (Finn et al., 2021), emphasizing the need to understand the condition of remaining estuarine habitat. Estuary conservation efforts are intended to protect coastal municipalities and provide sufficient habitat for wildlife. Stability of plant communities within tidal marshes contributes to the ability of these habitats to adapt to disturbance (Holling, 1973). A barrier to understanding community stability, including within estuaries, is the lack of long-term data. In the absence of long-term monitoring, historical datasets can provide a ‘snapshot’ of species compositional variation over time. One such opportunity exists in the Fraser River estuary, British Columbia, Canada in an area called Ladner Marsh (Fig. 1). Despite large-scale industrialization and urbanization within the region, Ladner Marsh has escaped direct industrial development, and to the best of our knowledge has not experienced major anthropogenic disturbance such as diking or agriculture in the past 50 years.

Two historical studies conducted in Ladner Marsh (Bradfield & Porter, 1982; Denoth & Myers, 2007) used similar methods to document floristic diversity. Bradfield & Porter (1982) tested whether species dominating the community statistically characterized distinct sub-community assemblages within the marsh. Their analysis distinguished three assemblages, each dominated by a unique species: Sedge (*Carex lyngbyei* Hornem.), Fescue (*Schedonorus arundinaceus* (Schreb., formerly *Festuca arundinacea*) Dumort., nom. cons.), and Bogbean (*Menyanthes trifoliata* L.). They postulated that edaphic factors drove assemblage distribution: that the Bogbean assemblage occurred on waterlogged soils, the Fescue assemblage on well-drained soils mostly along levees, and the Sedge assemblage along channel edges with greater inundation frequency. Twenty years later, Denoth & Myers (2007) repeated the sampling methods to test relationships between non-native purple loosestrife (*Lythrum salicaria* L.) and native Henderson’s checker-mallow (*Sidalcea hendersonii* S. Watson), a threatened species. While these studies independently characterize different community metrics, these datasets provide the opportunity to repeat observations and characterize long-term plant community changes to inform inferences about habitat stability. We used three observational datasets spanning four decades to answer the following questions:

1. How have assemblages in Ladner Marsh changed over the past 40 years? We would expect substantial changes in composition and abundance of species dominating assemblages to offer clues of processes driving change.
2. Are assemblages characterized by similar indicator plant species? If not, what species changes are associated with each assemblage? We expect that increasing abundance of non-native species over time would result in a greater net loss of native species.
3. Is the mean species diversity (α-diversity) and variation (β-diversity) within and between assemblages constant between the three sampling periods (1979, 1999, 2019)? If the plant community is stable, we expect little change in α-diversity and β-diversity.

# Methods

## Site history & context

The Fraser River is the largest watershed catchment in British Columbia, covering one quarter of the province (Finn et al., 2021). The current extent of the Fraser River Estuary spans 2,814 ha, one-third of which lies within the South Arm Marshes Wildlife Management Area, which was formally protected in 1991 (Schaefer, 2004) (Fig. 1B). Ladner Marsh occupies approximately 100 ha within the South Arm Marshes, bounded to the east by urban and industrial development and to the west by the Fraser River (Fig. 1).

Plant species common to these habitats are generally herbaceous, and the community is largely dominated by sedges, grasses, rushes, with a diversity of herbaceous flowering species (hereafter, forbs). Patterns of species distributions within tidal marshes are driven by elevation gradients, which filter species according to inundation and salinity regimes (Bertness & Ellison, 1987). The areas surveyed in Ladner Marsh (with the exception of Transect Q, omitted as explained in section ‘Differences between datasets’) correspond to elevations between the mean high water (MHW, approx. 1.1 m above mean sea level) and mean higher high water (MHHW, approx. 1.4 m above mean sea level) (as measured in Lane, 2022). Many species encountered in the surveys conducted are not restricted to these elevation ranges; emergent vegetation begins at the local mean tide (0 m above mean sea level), extending to the upper limit of tidally influenced inundation (Janousek et al., 2019). Thus, the elevation range of surveyed areas that we compare here occurred within a sufficiently restricted tidal elevation range that we do not expect elevation gradients and related hydrologic/salinity regimes to be a strong driver of species distributions within the areas surveyed.

## Sampling design & harmonization between observations

Our main goal was to sample the vegetation in a representative way to allow comparison with the datasets collected in 1979 (Bradfield & Porter, 1982) and 1999 (Denoth & Myers, 2007).

In the original 1979 study, eight transects (Q-X) were laid out in a subjective fashion to cross through the main features of vegetation diversity at Ladner Marsh (Fig. 1 in Bradfield & Porter, 1982). All transects spanned a similar elevation range across the marsh platform, with the three main plant assemblages (Sedge, Fescue, Bogbean) separated by apparent changes in hydrological conditions along transects.

In the 1999 study, Bradfield & Porter’s (1982) Fig.1 was used to visually approximate the locations of transects to repeat the vegetation survey (Denoth & Myers, 2007). In this study (2019 survey), transect locations were determined by overlaying Bradfield & Porter’s (1982) Fig. 1 on a georeferenced basemap, aligning prominent features such as tidal channel tributary junctions, marking GPS locations in Avenza Maps (Avenza Systems Inc., Ontario, Canada, v. 3.2), and finding these points in the field (Fig.1C). Difficulties arising from the inexact relocations of transects in the 1999 and 2019 surveys, and aggressive shrub encroachment along transect Q, resulted in an incomplete resampling of all eight transects from the original 1979 survey (further explained below). To evaluate the potential for differences in transect relocation to affect trends observed in the data, or to evaluate marsh-wide spatial trends in plant composition, we calculated the percentage of plots clustered in each assemblage group for each transect.

All three studies used a semi-systematic approach for determining locations of 1x1 m quadrats along transects. In the 1979 study, quadrats were mainly located at 10 m intervals along transects although this varied in places from 2-20 m depending on local changes in the vegetation (Bradfield & Porter, 1982). In the 1999 study, an attempt was made to follow the quadrat spacing shown in Bradfield & Porter’s (1982) Fig. 3 regardless of perceived vegetation changes along transects. For the 2019 study, quadrat placement was guided by visual assessment of vegetation patchiness along transects. If patches dominated (>50 % cover) by one or two species (not necessarily the three assemblage identifiers) continued more than 10 m of transect length, or if no dominant species was evident, we sampled every 10 m of transect length (Fig. 2D). No patches were so small that the 1 m2 plot was less than 1 m from the boundary of the next patch. Such fine-scale variations in decisions for quadrat placement among the three studies were considered inconsequential for the broader scale assessments of assemblage changes over time.

## Plot-scale sampling

Species were recorded if their most basal stem originated within the 1 m2 quadrat, and cover within the plot was considered for all above-ground vegetation that occurred within the quadrat boundary; vegetation overhanging the quadrat from an individual with its basal stem originating outside the quadrat boundary were not considered. In the instance where the basal stem was inside the plot, but aerial vegetation extended beyond the boundary of the quadrat, we only considered vegetation cover for portions of the plant within the boundary of the quadrat. We treated each ramet of rhizomatous species such as *Carex lyngbyei* or *Juncus* sp. as individuals, rather than attempting to delineate extent of each continuous rhizome of the genetically distinct individual. For these species, whenever the quadrat fell on top of an individual ramet, the ramet was considered in the plot if more than 50% of the leaves emerging from the ramet were immediately under or inside the quadrat boundary. Aerial plot cover was estimated by modified Braun-Blanquet cover classes used by Bradfield & Porter (1982) and Denoth & Myers (2007), where cover class 0 = 0% cover (absent), cover class 1 represents < 25% cover, cover class 2 represents 25-50% cover, cover class 3 represents 50-75% cover, and cover class 4 represents > 75% cover. Owing to consultation with one of the co-authors (Gary Bradfield) in the 1999 and 2019 studies, differences in between-observer cover estimation were considered minimal.

## Vegetation identification

For all sampling years, observation of vascular plant species was conducted in early summer when species are identifiable by sexual reproductive traits, but before senescence (approx. June – July). In all datasets, most plants were identified to species according to Hitchcock & Cronquist (1973), although a few were identified at higher taxonomic levels due to insufficient identifying characteristics (n = 6 to genus, n = 2 to Family; see Table S7). To account for changes in nomenclature revision over time, all datasets were harmonized to use the most recently accepted species name as reported in the PLANTS Database of the United States Department of Agriculture, Natural Resources Conservation Science [USDA NRCS]. For example, in the instance of *Agrostis* species, we assumed *Agrostis alba* L. identified in 1979 and 1999 was synonymous with *Agrostis stolonifera* L. in 2019. All species and their synonymous nomenclature from prior data collection years are available in Supplemental Table S7.

We elected to classify *Phalaris arundinacea* as non-native to align our treatment of the species with the designation provided by the British Columbia Ministry of Environment Species & Ecosystems Explorer (B.C. Conservation Data Center, 2023), which is the authoritative source for species conservation information for the province. While molecular analysis has confirmed *P. arundinacea* was native to North America prior to European colonization (Anderson et al., 2021), regional pollen studies have demonstrated some evidence for its absence in wetlands around the Salish Sea (Townsend & Hebda, 2013). Perhaps most important to consider is that hybridization of native with introduced varieties have resulted in aggressive invasive attributes, resulting in this species being of high management concern in Salish Sea Ecosystems (Sinks et al., 2021).

## Differences between datasets

In 1999 and 2019, some plots were omitted due to access or relocation issues (Table S1). Most notably, transect “Q” (n = 7 plots) was omitted in 1999 and 2019 due to inaccessibility. In 1979, this transect was placed within approx. 100 m of Ferry Rd, which forms the eastern boundary of the marsh by an approx. 2 m elevated grade to keep the road above high tide elevations. In this portion of the marsh, riparian forest with a dense understory of non-native Himalayan blackberry (*Rubus armeniacus* Focke) grew so densely that by 2019, access to the transect would have required significant and costly vegetation removal to access the area. The encroachment of blackberry and riparian thicket were also a challenge to surveyors in 1999 and were similarly omitted. Thus, data from transect Q is not included in the present analyses.

The 1999 survey approximately located all plots from transects R-X from the original 1979 survey; however, the number of plots along these transects differed in 2019. This is partially due to the surveyors in 1999 seeking to exactly relocate original plot locations, while in 2019 our objective was to place the plots according to visual perceptions of shifts in dominant species. Besides the plots omitted by not sampling transect Q, we noted a total of 20 fewer plots surveyed in 2019 (Table S1). This is most likely due to our methods in 2019 placing plots to characterize patches dominated by distinct species, resulting in number of plots being contingent on vegetation composition rather than spatial accuracy. Additionally, we acknowledge that spatial inaccuracy of transect relocation would result in different total transect lengths, and thus a different number of plots to be sampled along the transect. We also speculate there may have been some bank erosion resulting in wider channel mouths where some transects originated or ended, resulting in shorter transects overall. Visual comparison of satellite imagery suggests that erosion would have been minor, but not absent. To reconcile these differences, we excluded 1-4 plots per transect from the 1979 and 1999 datasets that had the least potential for spatial proximity to plots sampled in 2019 in order to compare an equal number of plots between sampling years along a similar length of transect (Table S1).

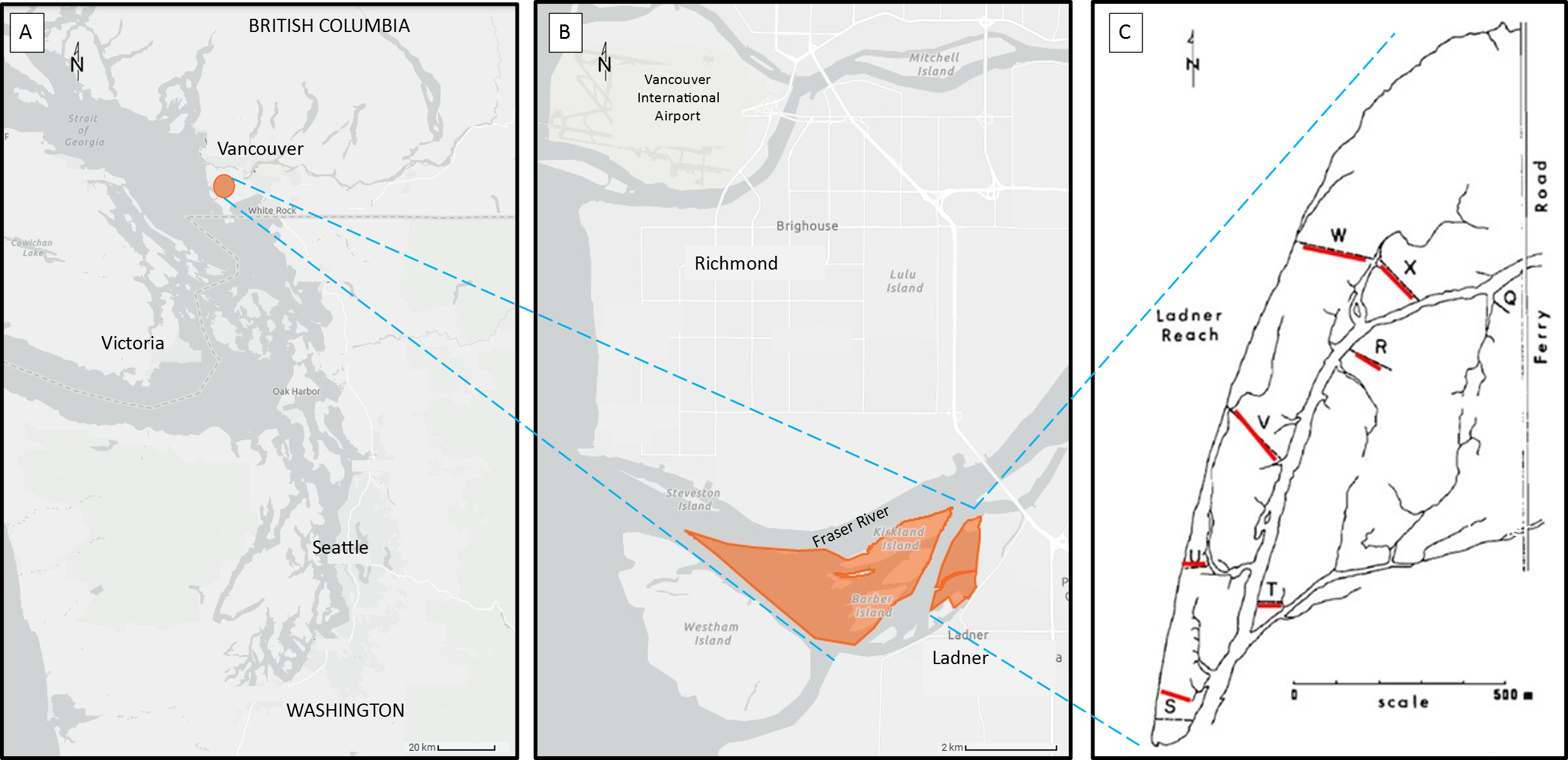
## Analyses

All analyses were performed in R v. 4.2.1 (R Core Team, 2022). We performed cluster analysis on species composition and abundance at the plot scale for each dataset. We used Euclidean distance as the measure of plot dissimilarity (“stats,” R Core Team) to facilitate direct comparisons to results produced by Bradfield & Porter (1982). Following (Legendre & Legendre, 2012), we also performed cluster analysis using Bray-Curtis dissimilarity to compare with Euclidean distance and found no meaningful difference in results from the two distance measures (Fig. S1).

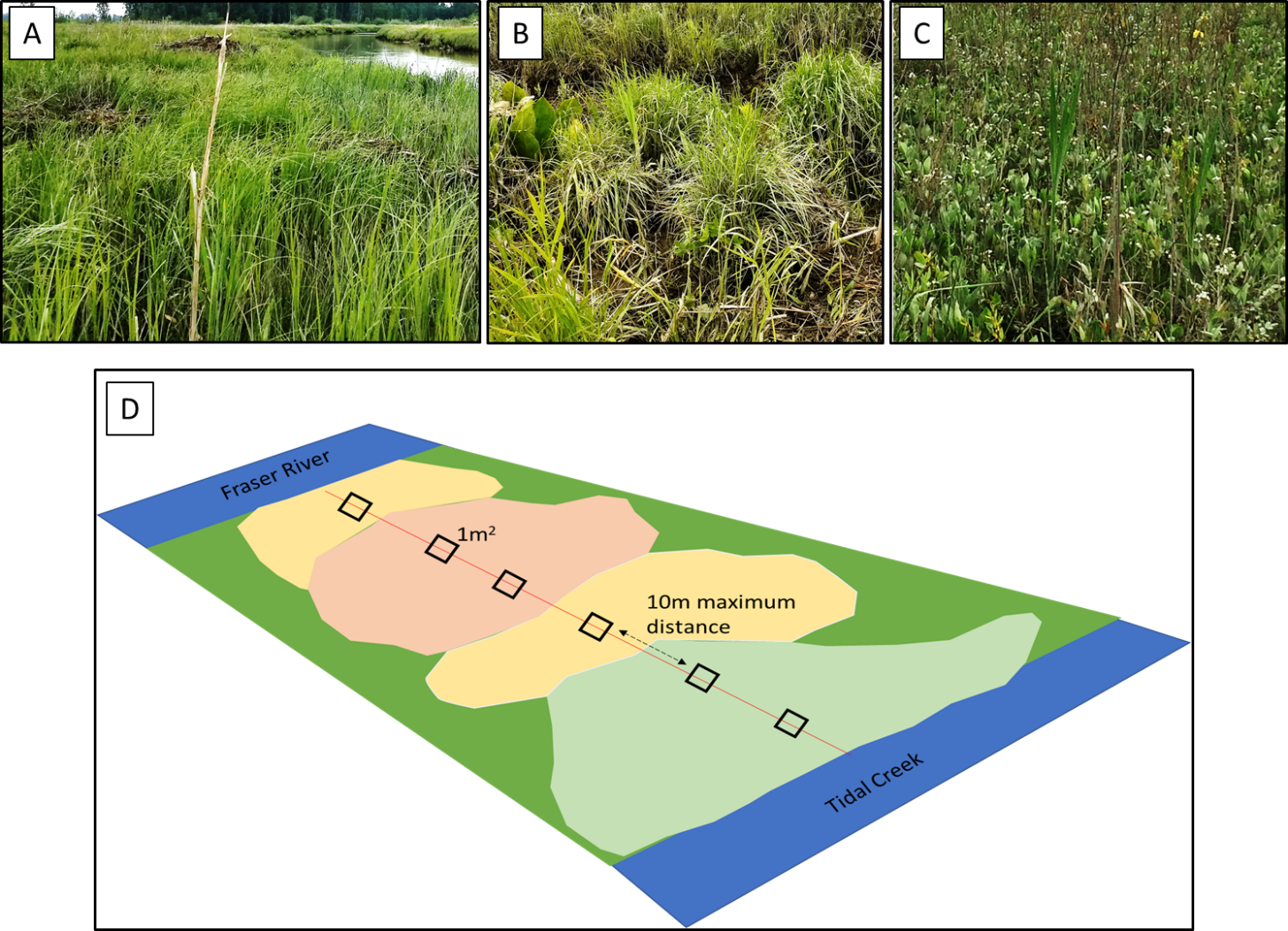
For each dataset, we identified three assemblages defined by the three highest cluster break points to facilitate direct comparisons of changes in vegetation properties over time. Species indicator analysis was used to determine which species abundance characterized each assemblage (“indicspecies,”R package De Cáceres & Jansen, 2016). Indicator Value (IndVal) association indices between species and clustered assemblages were calculated using an abundance-based point biserial correlation coefficient (multipatt func = “r.g”), and significance of associations was tested by permutational analysis (Dufrêne & Legendre, 1997). We also performed indicator analysis Bray-Curtis clusters to confirm the most significant indicator species were comparable between the two distance measures (Table S2). All species’ mean cover abundance is summarized in Table S6.

Community diversity calculations for each year of observation followed Whittaker (1975), with α-diversity calculated as the mean number of species per plot within an observation year and assemblage, and β-diversity calculated as the total number of species within the assemblage divided by α-diversity. These calculations were also performed on all data recorded for each observation year to generate community-wide measures of diversity. To address inconsistent numbers of plots grouped into assemblages each year, diversity metrics were bootstrapped 10 times using the minimum number of plots observed in an assemblage each year (n = 18) (Table S3).

Community turnover for each assemblage was measured using the “codyn” R package (Hallett et al., 2016). Total species turnover (total magnitude of change), species gained (appearances), and species lost (disappearances) were calculated as a percent change for each assemblage between 1979–1999, and 1999–2019. Total turnover was calculated as a ratio of the absolute value of species gained and lost to the total number of species observed in both timepoints.



**Fig. 1** Study area location and sampling design. (A) Regional location of the Fraser River Estuary in southwestern British Columbia, Canada, (B) South Arm Marshes Wildlife Management Area (highlighted in orange), (C) Ladner Marsh with overlay of 2019 transect locations (shown in red) on original transect map from Bradfield and Porter (1982). Base maps (A, B) generated by iMap published by the B. C. Conservation Data Center (Victoria, BC, Canada, <https://maps.gov.bc.ca/ess/hm/imap4m>).



**Fig. 2.** Dominant community vegetation characteristics observed in the (A) Sedge, (B) Fescue), and (C) Bogbean assemblages. (D) illustration of semi-systematic plot placement along transect (red line). At least one 1 m2 plot (black square) was placed within vegetation patches dominated by one or two species (multicolored polygons). Distance between plots varied, with minimum 1 m and maximum 10 m between all plots, regardless of the dominant species identified.

# Results

Three main assemblages identified by cluster analysis, characterized by the same dominant indicator species – Sedge (*Carex lyngbyei*), Fescue (*Schedonorus arundinaceus*), and Bogbean (*Menyanthes trifoliata*) – were evident across all sampling periods (Fig. 3). The three clusters formed at progressively lower Euclidean distance levels for the three sampling periods suggesting that assemblages were becoming more homogeneous with time. While the three assemblage indicator species remained constant over time, changes were evident in other species with significant indicator values (Table 1). For example, in 1979 the indicator species defining the Sedge assemblage cluster were *C. lyngbyei, Sagittaria latifolia* Wiild.*,* and *Schoenoplectus tabernaemontani* (C.C.Gmel.) Palla. In 1999, however, the same assemblage included indicator species *C. lyngbyei,* and *Impatiens capensis* Meerb. By 2019, *C. lyngbyei* was the only indicator for this assemblage. Similarly, *S. arundinaceus* remained a common indicator species within the Fescue assemblage, but the assemblage lost four out of seven total indicator species between 1979–2019. *Menyanthes trifoliata* consistently characterized the Bogbean assemblage, however non-native *Mentha aquatica* was common to the 1999 and 2019 datasets. For the Fescue and Bogbean assemblages, the importance of the assemblages’ namesake species shifted from the highest to second highest indicator species in at least one year (Table 1), however, we elected to maintain these species names as defining the assemblage, as they are common to all years of observation. While the identities of the remaining indicator species changed, there was no strong trend of changes in clade, or potential difference for changes in ecological function based on a qualitative review of changing species identity.

Across the entire Ladner Marsh plant community, two to three species were lost from each sampling year following the 1979 survey (Table S6). Within every assemblage α-diversity (mean number of species per plot) decreased every observation year, while β-diversity (ratio of total species in the assemblage to α-diversity) increased each year for all assemblages (Table 2). For example, the Sedge community suffered the least loss of species and α-diversity across sampling years, although β-diversity increased as in other assemblages, indicating increasing variability in which species may be encountered within a given assemblage. The Fescue assemblage had the greatest loss of α-diversity (> 50%) between 1979 and 2019. Approximately 1/3 fewer plots clustered as Fescue in 2019 than in 1979, however bootstrapping 18 random plots from every sampling year showed the same trend, indicating that loss of species was not related to loss of plots (Table S3). Total magnitude of species turnover between 1999 and 2019 was ~50% in each assemblage, largely driven by greater species disappearance (loss) between 1999 and 2019 (Table S4). Evaluation of spatial trends in assemblage changes across the marsh point to some potential for shifts to have been caused by inexact transect relocation (Fig. S2). For example, consistent percentage of plots within each assemblage group for transects W and X support relative accuracy in transect relocation between observers as well as plant assemblage stability. Variable patterns in percentage of plots in each assemblage such as the Bogbean assemblage on transects U and V may be indicative of spatial differences in transect relocation and/or greater turnover in a given year such that plots clustered into different assemblages.

The greatest loss of native species richness occurred in the Fescue assemblage, while gains in non-native richness were found in all assemblages (Fig. S3). The Fescue assemblage had a net loss of 17 native species between 1979 and 2019 (Table S5). Among the species lost from the Fescue assemblage, 12 were lost from all three assemblages (six forbs, six graminoids), or were never found in any other assemblage. Species gained include two woody species, and one each of forb, graminoid, and fern ally (*Equisetum arvense* L.). There was a net loss of one non-native species in the Fescue assemblage, however non-native, invasive *Phalaris arundinacea* (reed canary grass) accounts for the greatest 2019 mean cover in the entire assemblage (25–50% mean cover, Table S5). In the Bogbean assemblage, the net gain of two non-native species included *P. arundinacea* and *Iris pseudacorus* L. (yellow flag iris). Within the Sedge assemblage, there was a net loss of two native species, and net gain of two non-native species, including *P. arundinacea* and *I. pseudacorus*. As of 2019, these species accounted for < 25% mean cover, but may be of significant management concern (Fig. 4).

Assemblage-defining indicator species showed an overall trend of decreasing cover over time (Fig. 4). Notably, in the Fescue assemblage, the cover class of non-native indicator *S. arundinaceus* fell from a mean of ~1.5 to ~0.75 from 1979–2019, while the mean cover class of non-native *P. arundinacea* tripled from 1999–2019. In the Sedge assemblage native indicator sedge species *C. lyngbyei* decreased in cover from 1979–2019 (Fig. 4), stepping down from a mean cover class value of 3 (50–75% cover) to 2 (25–50% cover) between 1979–2019. Meanwhile, non-native species *L. salicaria* and *S. arundinaceus* increased in their mean cover abundance, although both species remained in the same mean cover class (< 25% mean cover) by 2019. Similarly, in the Bogbean assemblage, cover abundance of native species *M. trifoliata* declined from a mean cover class of 4 (> 75%) to 3 (50-75% mean cover) by 2019, while cover of non-native *Mentha aquatica* L. increased from a mean cover class of 0.4 in 1979 (Table S5) to a mean cover class of ~2 (~25-50% mean cover) by 2019 (Fig. 4, Table S5).





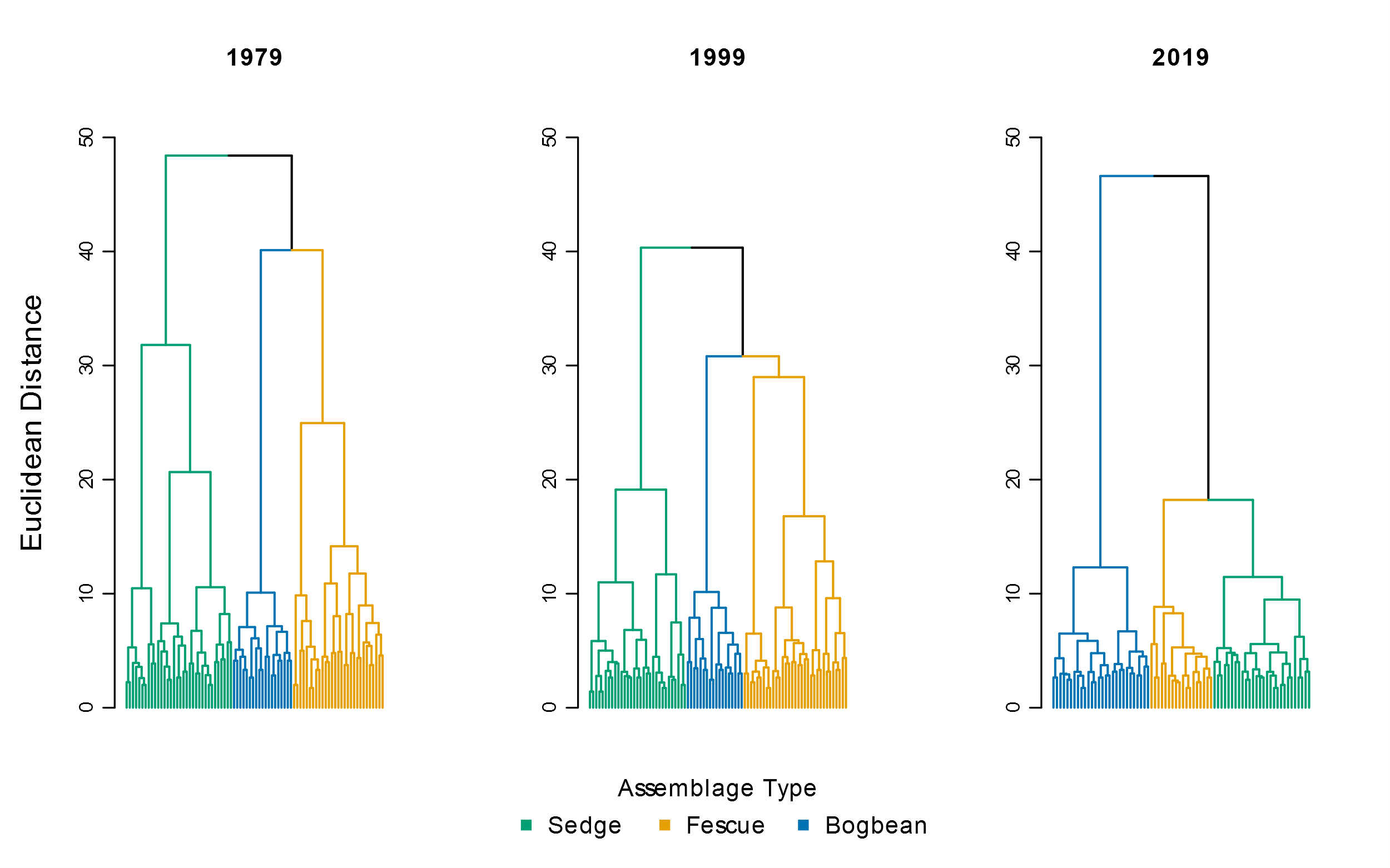


**Table 1** Species significantly driving cluster groups (Euclidean distance) include the same dominant species in each assemblage type (Sedge by *Carex lyngbyei*, Fescue by *Schedonorus arundinaceus*, Bogbean by *Menyanthes trifoliata*). Indicator species significantly defining the assemblage reported for p < 0.05

|  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | **1979** | | |  | **1999** | | |  | **2019** | | |
| Cluster Group Name | Species | Indicator Value | p-value |  | Species | Indicator Value | p-value |  | Species | Indicator Value | p-value |
|  |  |  |  |  |  |  |  |  |  |  |  |
| "Sedge" | *Carex lyngbyei* | 0.722 | 0.0001 |  | *Carex lyngbyei* | 0.626 | 0.0001 |  | *Carex lyngbyei* | 0.591 | 0.0001 |
| *Sagittaria latifolia* | 0.523 | 0.0001 |  | *Impatiens capensis* | 0.320 | 0.0147 |  |  |  |  |
| *Schoenoplectus tabernaemontani* | 0.417 | 0.0004 |  |  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |
| "Fescue" | *Schedonorus arundinaceus* | 0.607 | 0.0001 |  | *Poa palustris* | 0.569 | 0.0001 |  | *Phalaris arundinacea* | 0.518 | 0.0001 |
| *Salix lasiandra* | 0.535 | 0.0001 |  | *Schedonorus arundinaceus* | 0.399 | 0.0006 |  | *Schedonorus arundinaceus* | 0.461 | 0.0001 |
| *Equisetum palustre* | 0.489 | 0.0001 |  | *Trifolium wormskioldii* | 0.398 | 0.0014 |  | *Equisetum fluviatile* | 0.320 | 0.0127 |
| *Lathyrus palustris* | 0.433 | 0.0003 |  | *Bidens cernua* | 0.371 | 0.0044 |  |  |  |  |
| *Sidalcea hendersonii* | 0.331 | 0.0058 |  |  |  |  |  |  |  |  |
| *Hordeum brachyantherum* | 0.293 | 0.0157 |  |  |  |  |  |  |  |  |
| *Deschampsia caespitosa* | 0.267 | 0.0455 |  |  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |  |  |  |
| "Bogbean" | *Menyanthes trifoliata* | 0.729 | 0.0001 |  | *Mentha aquatica* | 0.811 | 0.0001 |  | *Menyanthes trifoliata* | 0.942 | 0.0001 |
| *Myosotis scorpiodes* | 0.446 | 0.0003 |  | *Menyanthes trifoliata* | 0.621 | 0.0001 |  | *Mentha aquatica* | 0.618 | 0.0001 |
| *Bidens cernua* | 0.407 | 0.0012 |  | Grass (unidentified) | 0.452 | 0.0005 |  | *Lysimachia thyrsiflora* | 0.537 | 0.0001 |
| *Lythrum salicaria* | 0.406 | 0.0012 |  | *Lythrum salicaria* | 0.424 | 0.0012 |  | *Galium trifidum* | 0.465 | 0.0006 |
| *Equisetum fluviatile* | 0.326 | 0.0106 |  | *Juncus articulatus* | 0.417 | 0.0005 |  | *Myosotis scorpioides* | 0.392 | 0.0056 |
| *Lysimachia thyrsiflora* | 0.321 | 0.0103 |  | *Equisetum fluviatile* | 0.404 | 0.0016 |  | *Juncus articulatus* | 0.334 | 0.0151 |
|  |  |  |  | *Myosotis scorpioides* | 0.352 | 0.0046 |  |  |  |  |
|  |  |  |  | *Eleocharis palustris* | 0.303 | 0.0224 |  |  |  |  |
|  |  |  |  | *Equisetum variegatum* | 0.277 | 0.0447 |  |  |  |  |
|  |  |  |  | *Deschampsia caespitosa* | 0.273 | 0.0270 |  |  |  |  |

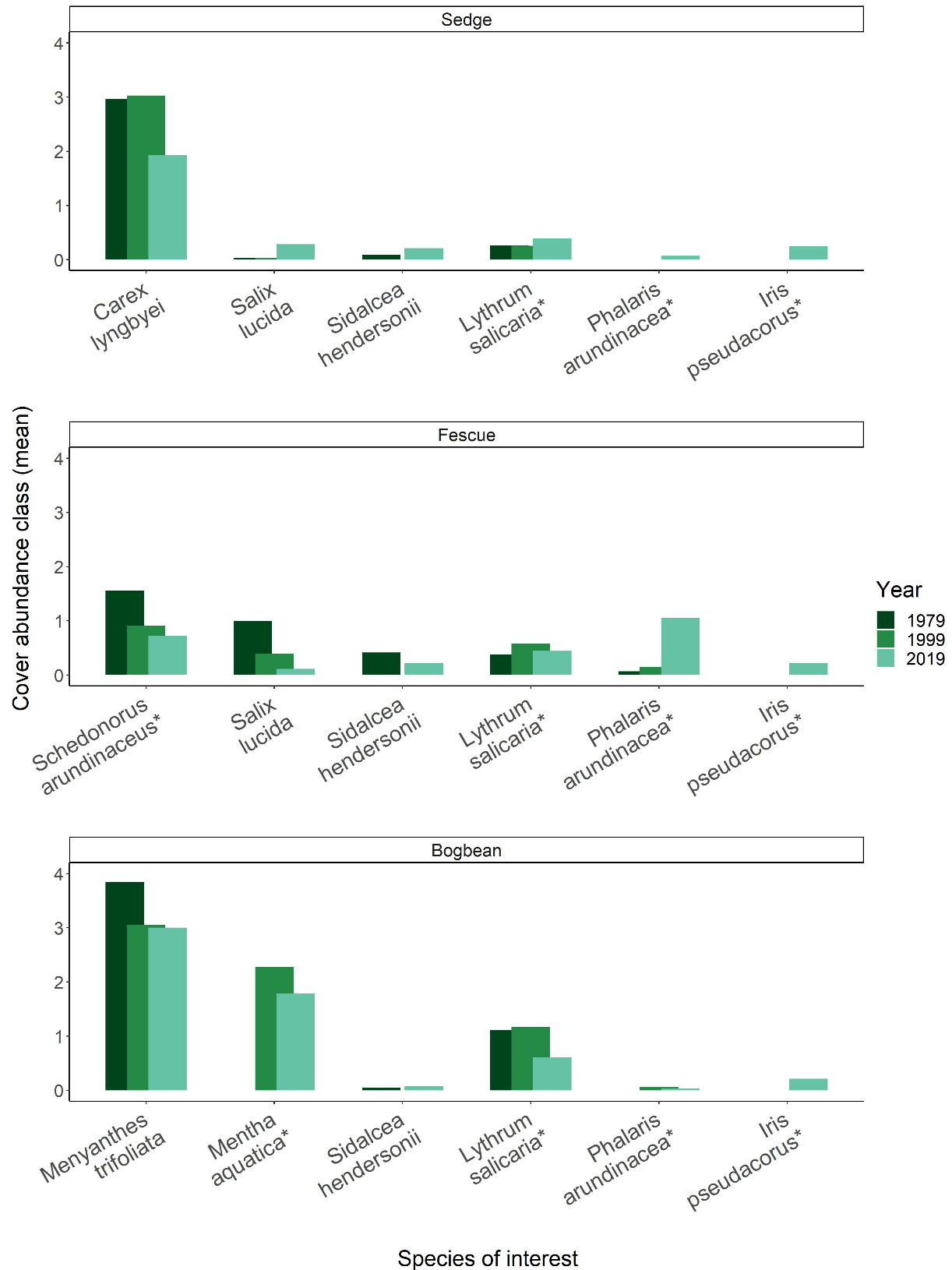
**2**Data compared between, 1999, surveys resulted infive over time

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  |  | |  |  | | |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |



**Fig. 3** Dendrograms from cluster analysis of species cover class data from each sampling period showing the three main assemblage types, defined by the three highest break points in the dendrogram. Euclidean distance from unity to three highest break levels increased in each dataset, especially in 2019 dataset, indicating greater dissimilarity between assemblages over time.

~~Species cover abundance becomes more dissimilar in each assemblage over time, as shown by greater Euclidean distance between assemblage types. Note clusters of the Sedge and Fescue assemblages are more similar in 2019~~



**Fig. 4** Changes in mean cover abundance (cover classes) for select significant indicator species (*Carex lyngbyei, Schedonorus arundinaceus, Menyanthes trifoliata*), the most-abundant woody species (*Salix lucida*), and native/non-native species of local management interest (*Sidalcea hendersonii, Lythrum salicaria, Phalaris arundinacea, Iris pseudacorus*). Non-native species denoted by asterisk (\*). Significant indicator species within each assemblage have decreased in abundance over time, while several non-native species have increased in cover abundance since 1979. Cover classes are: [1] = < 25%, [2] = 25-50%, [3] = 51-75%, [4] = >75% above-ground vegetated cover.

# Discussion

Despite its status as a Wildlife Management Area and general resilience of the Fraser River tidal marsh ecosystem we found substantive changes in species composition over a 40-year time-frame, potentially indicating broader-scale processes affected by regional pressures. The three species significantly characterizing the three plant assemblages, Sedge (*C. lyngbyei*), Fescue (*S. arundinaceus*) and Bogbean (*M. trifoliata*), have remained the same over the past 40 years. We observed, however, a decline of native species richness accompanied by an increased richness and abundance of non-native species, including invasive non-native species. Of some concern is our observation of the homogenization of cover abundance within assemblages, and overall loss of indicator species for the Sedge and Fescue assemblages. Increasing abundance of non-native species within each assemblage is likely driving the greater similarity within assemblages (homogenization) and greater dissimilarity between assemblages, as shown by cluster analysis (Fig. 3). While addition of non-native species can contribute to greater biodiversity (Sagoff, 2005), the homogenization of plant communities (especially by dominance of non-native invasive species) leads to lower diversity overall (Houlahan & Findlay, 2004), which in turn may lead to lower functional redundancy and potential for reduced ecosystem stability (de Bello et al., 2021).

The changing identity of species or functional traits in an assemblage may offer clues to shifting abiotic conditions within or between assemblages (Waller et al., 2020). One functional group to note were the woody species, as their traits convey different structural habitat qualities than herbaceous species. Willow (*Salix* *lucida* Muhl.) was most prevalent in the Fescue assemblage in 1979, but was most abundant in the Sedge assemblage in 2019. This could suggest long-term shifts in edaphic factors, nutrient regimes, and/or the competitive encroachment of invasive reed canary grass (*Phalaris arundinacea*), making the Fescue assemblage less hospitable to willow recruitment. Alternatively, this could indicate that environmental conditions are becoming more similar between the two assemblages, as evidenced by the clustering of the Fescue and Sedge groups on the same branch in the 2019 dendrogram (Fig. 3). The indicator species analysis for the Sedge assemblage in 1979 included plants tolerant of highly saturated soils (*Sagittaria latifolia, Schoenoplectus tabernaemontani*), but in 1999 the assemblage indicators included species less tolerant of aquatic or constantly saturated soils (*Impatiens capensis*) (Table 1).

In contrast, the turnover of indicator species may simply represent variation in species compositional abundance in each sampling year, despite being a perennial-dominated community. For example, the Bogbean assemblage, was indicated largely by unique forbs in 1979 and 2019, and an even mix of unique forbs and graminoids in 1999 (Table 1). It is harder to attribute replacement of forb indicator species to potential woody riparian succession in the Bogbean assemblage as in the Sedge and Fescue assemblages. The indicator graminoid species found only in 1999 in the Bogbean assemblage (excluding an unknown grass identified only to family) are all native wetland species commonly found in brackish estuarine marshes in the Pacific Northwest of North America. Rather than indicating altered abiotic conditions, their inclusion as indicator species may simply represent population dynamics of short-lived perennials such as dispersal and recruitment. Thus, we propose two potential alternative explanations for the observed changes in floristic composition observed in the different assemblages: cumulative stressors such as altered nutrient regimes or hydrogeomorphological processes (e.g., altered sedimentation rates relative to tidal inundation) may be slowly altering abiotic conditions to favor different species within each of the assemblages identified. Alternatively, population dynamics or interspecific interactions (e.g., competition) may be operating independently of abiotic conditions, or have different outcomes depending on abiotic conditions in each assemblage. Testing how life histories (e.g., species longevity vs. recruitment) offer competitive advantage in the context of changing abiotic conditions would be a valuable long-term addition to general interactions of competition and edaphic factors. These interactions would present a valuable experimental test of competitive advantage or how edaphic conditions drive the dominance of native vs. non-native species in tidal wetlands.

Decreasing frequency of unique species within assemblages (Table 3), and decreasing similarity of compositional abundance between assemblages (Fig. 3), may result from overall loss of native floristic richness. Across all assemblages in Ladner Marsh 1979–2019, we found one to two fewer native species, while β-diversity increased. This would indicate that rare (infrequently found) species are becoming more locally rare, which contributes to the loss of heterogeneous cover abundance and increased β-diversity observed at the plot scale. More concerning is the net loss of five perennial graminoid and forb species over the study period (Table S6), as this potentially represents a loss of functional redundancy. This species loss from the observed datasets may not represent species loss from the entire Ladner Marsh Wildlife Management Area. Nonetheless, the net species loss from the dataset, along with the addition of three non-native species to the datasets, poses concern for potential of species loss from the habitat over time.

Plant biodiversity loss may have consequences such as altered sediment trapping via altered vegetation structural complexity (Bouma et al., 2010) or reduce availability of important pollinator plants (Newbold et al., 2019). However, these contributions by the species lost in Ladner Marsh have not been quantified. Regardless of whether the loss is due to turnover or shifting abiotic conditions, trends of lost native plant species richness may indicate greater susceptibility to invasion (Kuiters, *et al.*, 2009), and thus a loss of resistance to non-native species encroachment over time. This can be evidenced by the decreasing ratio of native to non-native cover across Ladner Marsh 1979–2019 (Fig. S3), although few species (native or non-native) represent the majority of cover within the assemblage (Table S5). Although non-native species of significant management concern (e.g., *P. arundinacea*, *I. pseudacorus*)) were < 25% mean plot cover in 2019, these species are notorious for spreading to the point of near-exclusion of other species (especially natives) (Apfelbaum & Sams, 1987; Sinks et al., 2021).

## Mechanisms, Synthesis & Recommendations

Non-native species invasion and native species loss may lead to instability in native populations through fragmented or lost propagule dispersal networks, potentially leading to ecosystem instability through altered trophic cascades (Duffy, 2003). Disentangling explicit effects of abiotic processes of sedimentation, propagule dispersal, or propagule recruitment from other biotic interactions would be no easy task in a tidal marsh ecosystem; however, experimentally testing optimal recruitment niches of species-specific propagules (e.g., Lane, 2022) could prove valuable for understanding best practices to maintain at-risk populations or test community function.

Optimal abiotic conditions for the recruitment and spatial occupancy of native or non-native species may largely be driven by soil characteristics and related sedimentation processes. Changes such as sediment starvation, subsidence, or relative sea level rise would result in more saturated areas, which would likely drive the increased prevalence of saturated conditions favored by the Bogbean assemblage (Mendelssohn & Kuhn, 2003). Alternatively, positive feedbacks between vegetation and sedimentation could support areas of marsh accretion (Nyman et al., 2006), which may also be more likely to receive non-native propagules within the distributed sediment. While Ladner Marsh has largely escaped direct anthropogenic disturbance (e.g., industrial or agricultural development), it is subject to continuous pressures resulting from anthropogenic modifications throughout the Fraser River Estuary. Cumulative effects of altered water, sediment, and nutrient regimes impacting the lower reaches of the Fraser River can alter competitive dynamics of plant communities (Dethier & Hacker, 2005; Flores-Moreno et al., 2016), and promote the dominance of invasive species (Green & Galatowitsch, 2002; Woo & Zedler, 2002; Zedler & Kercher, 2004). In turn, this may facilitate dispersal and recruitment of non-native species and potentially limit the dispersal and recruitment of native species because propagule pools are dependent on local and regional proximity. If similar habitats within tidal estuarine ecosystems are lost to the point where distance between patches exceeds propagule dispersal distance (Shi, et al., 2020), then species colonization within the ecosystem is rare or lost (but see Stewart et al., 2022). Alternatively, if non-native species are more prevalent throughout the regional dispersal network, then there is a greater chance of non-native species introduction within a local marsh community (Briski et al., 2012).

A common (mis)assumption is that “undisturbed” protected areas such as Ladner Marsh represent ecologically appropriate reference states (e.g., Stoddard, et al., 2006, and citations therein). Our findings illustrate how, in a heavily impacted region (Finn et al., 2021), compositional states have likely shifted from recent (< 100 years) historical references, yet may still contribute value as an example of potential ecological benchmarks for restoration success (Shackelford, et al., 2021). However, the designation of Ladner Marsh as a Wildlife Management Area is likely insufficient to protect the habitat from large-scale environmental stressors in the Fraser River Estuary, such as nutrient enrichment. We suggest that the plant community changes described here should alert land managers not only to what species diversity might be targeted in conservation practice, but also to how reference sites may have changed with respect to non-native, invasive encroachment during the span of 20–40 years. We strongly advocate for the development of long-term vegetation monitoring to inform non-native invasive species management occurring in this and similar WMAs (see also Stewart, Hood, and Martin, 2023).

If we are to prioritize conservation of functional coastal wetlands that include a significant representation of native species, we must seek new ways to manage habitats such as Ladner Marsh. Active management may be required to maintain ecologically-desired species composition in the wake of environmental change, and should be informed by ongoing experimentation into the role of hydrogeomorphologic drivers, dispersal networks, recruitment strategies, disturbance, and invasive species management to achieve this goal. In so doing, practitioners may enhance ecosystem processes within remnant coastal wetland habitats. This active management process also presents a timely and necessary opportunity in the Pacific Northwest of North America to engage with First Nations to revive traditional management practices in tidal wetlands, such as select mechanical disturbance (Turner, 2014). Working with traditional knowledge holders in these ecosystems may yield deeper understanding of plant community function and habitat stability, which would enhance ecosystem resilience and potentially lead to positive outcomes on regionally important salmonid and shorebird populations while contributing to reconciliation between Indigenous and colonial cultures.

# Statements & Declarations

## Funding

Financial support for 2019 field surveys was provided by Natural Sciences and Engineering Research Council of Canada Discovery Grant RGPIN-2018-03838 to John S. Richardson (University of British Columbia). Analysis and manuscript writing was supported by Liber Ero Chair in Conservation to Tara G. Martin, and a Mitacs Accelerate Fellowship to Stefanie L. Lane.

## Competing interests

The authors have no relevant financial or non-financial interests to disclose.

## Author contributions

Study conception, 2019 data collection, analysis, and interpretation were undertaken by Stefanie L. Lane. Gary E. Bradfield oversaw original study design and publication (Bradfield & Porter, 1982), as well as supporting Methods comparisons between datasets and evaluating Results presented in this study. Madlen Denoth contributed data collected in 1999. Nancy Shackelford assisted with theoretical framework and manuscript revision. Manuscript was drafted by Stefanie L. Lane; Gary E. Bradfield, Nancy Shackelford, and Tara G. Martin supported draft revisions on all previous versions of this manuscript. All authors read and approved the final manuscript.

## Data availability

Data and code for all years of observation are available on GitHub (<https://github.com/stefanielane/CommunityStability.git>), or via Dryad (<https://doi.org/10.5061/dryad.r7sqv9sh8>)

# Literature Cited

Anderson, N. O., Smith, A. G., Noyszewski, A. K., Ito, E., Dalbotten, D., & Pellerin, H. (2021). Management and control issues for native, invasive species (reed Canarygrass): Evaluating philosophical, management, and legislative issues. *HortTechnology, 31*(4) 354-366.

Apfelbaum, S. I., & Sams, C. E. (1987). Ecology and Control of Reed Canary Grass (Phalaris arundinacea L.). *Natural Areas Journal*, *7*(2), 69–74.

Bai, Y., Han, X., Wu, J., Chen, Z., & Li, L. (2004). Ecosystem stability and compensatory effects in the Inner Mongolia grassland. *Nature*, *431*(7005), 181–184. <https://doi.org/10.1038/nature02850>

B.C. Conservation Data Centre (2023). BC Species and Ecosystems Explorer. B.C. Minist. of Environ. Victoria, B.C. Available: <https://a100.gov.bc.ca/pub/eswp/> (accessed Sep 18, 2023).

Bertness, M. D., & Ellison, A. M. (1987). Determinants of Pattern in a New England Salt Marsh Plant Community. Ecological Monographs, 57(2), 129-147.

Bouma, T. J., Vries, M. B. De, & Herman, P. M. J. (2010). Comparing ecosystem engineering efficiency of two plant species with contrasting growth strategies. Ecology 91(9), 2696-2704.

Bradfield, G. E., & Porter, G. L. (1982). Vegetation structure and diversity components of a Fraser estuary tidal marsh. Canadian Journal of Botany, 60(4), 440–451. https://doi.org/10.1139/b82-060

Briski, E., Bailey, S. A., Casas-Monroy, O., DiBacco, C., Kaczmarska, I., Levings, C., ... & MacIsaac, H. J. (2012). Relationship between propagule pressure and colonization pressure in invasion ecology: a test with ships' ballast. Proceedings of the Royal Society B: Biological Sciences, 279(1740), 2990-2997.

Brophy, L. S., Greene, C. M., Hare, V. C., Holycross, B., Lanier, A., Heady, W. N., O’Connor, K., Imaki, H., Haddad, T., & Dana, R. (2019). Insights into estuary habitat loss in the western United States using a new method for mapping maximum extent of tidal wetlands. *PLOS ONE*, *14*(8), e0218558. https://doi.org/10.1371/journal.pone.0218558

Bruno, J. F. (2000). Facilitation of Cobble Beach Plant Communities Through Habitat Modification by Spartina Alterniflora. *Ecology*, *81*(5), 1179–1192. https://doi.org/10.1890/0012-9658(2000)081[1179:FOCBPC]2.0.CO;2

Butzeck, C., Schröder, U., Oldeland, J., Nolte, S., & Jensen, K. (2016). Vegetation succession of low estuarine marshes is affected by distance to navigation channel and changes in water level. *Journal of Coastal Conservation*, *20*(3), 221–236. https://doi.org/10.1007/s11852-016-0432-1

Chalifour, L., Scott, D. C., MacDuffee, M., Iacarella, J. C., Martin, T. G., & Baum, J. K. (2019). Habitat use by juvenile salmon, other migratory fish, and resident fish species underscores the importance of estuarine habitat mosaics. *Marine Ecology Progress Series*, *625*, 145–162. https://doi.org/10.3354/meps13064

Davis, M. J., Woo, I., Ellings, C. S., Hodgson, S., Beauchamp, D. A., Nakai, G., & De La Cruz, S. E. W. (2021). A climate-mediated shift in the estuarine habitat mosaic limits prey availability and reduces nursery quality for juvenile salmon. *Estuaries and Coasts*. https://doi.org/10.1007/s12237-021-01003-3

de Bello, F., Lavorel, S., Hallett, L. M., Valencia, E., Garnier, E., Roscher, C., Conti, L., Galland, T., Goberna, M., Májeková, M., Montesinos-Navarro, A., Pausas, J. G., Verdú, M., E-Vojtkó, A., Götzenberger, L., & Lepš, J. (2021). Functional trait effects on ecosystem stability: Assembling the jigsaw puzzle. *Trends in Ecology & Evolution*, *36*(9), 822–836. https://doi.org/10.1016/j.tree.2021.05.001

De Cáceres, M., & Jansen, F. (2016). *Indicspecies*. http://r.meteo.uni.wroc.pl/web/packages/indicspecies/indicspecies.pdf

Denoth, M., & Myers, J. H. (2007). Competition between Lythrum salicaria and a rare species: Combining evidence from experiments and long-term monitoring. *Plant Ecology*, *191*(2), 153–161. https://doi.org/10.1007/s11258-006-9232-2

Dethier, M. N., & Hacker, S. D. (2005). Physical factors vs. biotic resistance in controlling the invasion of an estuarine marsh grass. Ecological Applications, 15(4), 1273-1283. https://doi.org/10.1890/04-0505

Donohue, I., Hillebrand, H., Montoya, J. M., Petchey, O. L., Pimm, S. L., Fowler, M. S., Healy, K., Jackson, A. L., Lurgi, M., McClean, D., O’Connor, N. E., O’Gorman, E. J., & Yang, Q. (2016). Navigating the complexity of ecological stability. *Ecology Letters*, *19*(9), 1172–1185. https://doi.org/10.1111/ele.12648

Douglas, T. J., Schuerholz, G., & Juniper, S. K. (2022). Blue Carbon Storage in a Northern Temperate Estuary Subject to Habitat Loss and Chronic Habitat Disturbance: Cowichan Estuary, British Columbia, Canada. *Frontiers in Marine Science*, *9*. https://www.frontiersin.org/article/10.3389/fmars.2022.857586

Duffy, J. E. (2003). Biodiversity loss, trophic skew and ecosystem functioning. *Ecology Letters*, *6*(8), 680–687. https://doi.org/10.1046/j.1461-0248.2003.00494.x

Dufrêne, M., & Legendre, P. (1997). Species Assemblages and Indicator Species: the Need for a Flexible Asymmetrical Approach. *Ecological Monographs*, *67*(3), 345–366. https://doi.org/10.1890/0012-9615(1997)067[0345:SAAIST]2.0.CO;2

Emmett, R., Llansó, R., Newton, J., Thom, R., Hornberger, M., Morgan, C., Levings, C., Copping, A., & Fishman, P. (2000). Geographic signatures of North American West Coast estuaries. *Estuaries*, *23*(6), 765–792. http://dx.doi.org/10.2307/1352998

Finn, R. J. R., Chalifour, L., Gergel, S. E., Hinch, S. G., Scott, D. C., & Martin, T. G. (2021). Quantifying lost and inaccessible habitat for Pacific salmon in Canada’s Lower Fraser River. *Ecosphere*, *12*(7), e03646. https://doi.org/10.1002/ecs2.3646

Flores-Moreno, H., Reich, P. B., Lind, E. M., Sullivan, L. L., Seabloom, E. W., Yahdjian, L., ... & Borer, E. T. (2016). Climate modifies response of non-native and native species richness to nutrient enrichment. Philosophical Transactions of the Royal Society B: Biological Sciences, 371(1694), 20150273. https://doi.org/10.1098/rstb.2015.0273

Gailis, M., Kohfeld, K. E., Pellat, M. G., & Carlson, D. (2021). Quantifying blue carbon for the largest salt marsh in southern British Columbia: implications for regional coastal management. *Coastal Engineering Journal*, 63(3), 275-309. https://doi.org/10.1080/21664250.2021.1894815

Hallett, L. M., Jones, S. K., MacDonald, A. A. M., Jones, M. B., Flynn, D. F. B., Ripplinger, J., Slaughter, P., Gries, C., & Collins, S. L. (2016). codyn: An r package of community dynamics metrics. *Methods in Ecology and Evolution*, *7*(10), 1146–1151. https://doi.org/10.1111/2041-210X.12569

Hanski, I. (1982). Dynamics of Regional Distribution: The Core and Satellite Species Hypothesis. *Oikos*, *38*(2), 210–221. JSTOR. https://doi.org/10.2307/3544021

Hitchcock, C. L., & Cronquist, A. (1973). *Flora of the Pacific Northwest, an illustrated manual*. University of Washington Press.

Holling, C. S. (1973). Resilience and Stability of Ecological Systems. *Annual Review of Ecology and Systematics*, *4*(1), 1–23. https://doi.org/10.1146/annurev.es.04.110173.000245

Houlahan, J. E., & Findlay, C. S. (2004). Effect of Invasive Plant Species on Temperate Wetland Plant Diversity. *Conservation Biology*, *18*(4), 1132–1138. https://doi.org/10.1111/j.1523-1739.2004.00391.x

Kehoe, L. J., Lund, J., Chalifour, L., Asadian, Y., Balke, E., Boyd, S., Carlson, D., Casey, J. M., Connors, B., Cryer, N., Drever, M. C., Hinch, S., Levings, C., MacDuffee, M., McGregor, H., Richardson, J., Scott, D. C., Stewart, D., Vennesland, R. G., … Martin, T. G. (2021). Conservation in heavily urbanized biodiverse regions requires urgent management action and attention to governance. *Conservation Science and Practice*, *3*(2), e310. https://doi.org/10.1111/csp2.310

Kopecký, M., & Macek, M. (2015). Vegetation resurvey is robust to plot location uncertainty. *Diversity and Distributions*, *21*(3), 322–330. https://doi.org/10.1111/ddi.12299

Lane, S. L. (2022). Using marsh organs to test seed recruitment in tidal freshwater marshes. *Applications in Plant Sciences*, *n/a*, e11474. https://doi.org/10.1002/aps3.11474

Legendre, P., & Legendre, L. (2012). *Numerical Ecology* (3rd ed., Vol. 24). Elsevier.

Lepš, J. (2004). What do the biodiversity experiments tell us about consequences of plant species loss in the real world? *Basic and Applied Ecology*, *5*(6), 529–534. https://doi.org/10.1016/j.baae.2004.06.003

Mendelssohn, I. A., & Kuhn, N. L. (2003). Sediment subsidy: Effects on soil–plant responses in a rapidly submerging coastal salt marsh. *Ecological Engineering*, *21*(2), 115–128. <https://doi.org/10.1016/j.ecoleng.2003.09.006>

Newbold, T., Adams, G. L, Albaladejo Robales, G., Boakes, E. H., Braga Ferreira, G., Chapman, A. S. A., Etard, A., Gibb, R., Millard, J., Outhwaite, C. L., Williams, J. J. (2019). Climate and land-use change homogenise terrestrial biodiversity, with consequences for ecosystem functioning and human well-being. Emerging Topics in Life Sciences, (3)2, 2017-219.

Nyman, J. A., Walters, R. J., Delaune, R. D., & Patrick, W. H. (2006). Marsh vertical accretion via vegetative growth. Estuarine, Coastal and Shelf *Science*, *69*(3), 370–380. https://doi.org/10.1016/j.ecss.2006.05.041

Ovaskainen, O., Rybicki, J., & Abrego, N. (2019). What can observational data reveal about metacommunity processes? *Ecography*, *42*(11), 1877–1886. https://doi.org/10.1111/ecog.04444

Palmer, M. A., Ambrose, R. F., & Poff, N. L. (1997). Ecological Theory and Community Restoration Ecology. *Restoration Ecology*, *5*(4), 291–300. https://doi.org/10.1046/j.1526-100X.1997.00543.x@10.1111/(ISSN)1526-100X.2525thAnniversaryVI

Pasternack, G. B. (2009). Chapter 3. Hydrogeomorphology and sedimentation in tidal freshwater wetlands. In A. Barendregt, D. F. Whigham, & A. H. Baldwin (Eds.), *Tidal Freshwater Wetlands* (pp. 31–40). Backhuys Publishers.

R Core Team (2022). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Sagoff, M. (2005). Do Non-Native Species Threaten The Natural Environment? *Journal of Agricultural and Environmental Ethics*, *18*(3), 215–236. https://doi.org/10.1007/s10806-005-1500-y

Schaefer, V. (2004). Ecological setting of the Fraser River delta and its urban estuary. In B. J. Groulx, D. C. Mosher, J. L. Luternauer, & D. E. Bilderback (Eds.), *Fraser River Delta, British Columbia: Issues of an Urban Estuary* (pp. 147–172). Geological Survey of Canada, Bulletin 547.

Shackelford, N., Dudney, J., Stueber, M. M., Temperton, V. M., & Suding, K. L. (2021). Measuring at all scales: Sourcing data for more flexible restoration references. *Restoration Ecology*, *n/a*(n/a), e13541. https://doi.org/10.1111/rec.13541

Shi, W., Shao, D., Gualtieri, C., Purnama, A., & Cui, B. (2020). Modelling long-distance floating seed dispersal in salt marsh tidal channels. *Ecohydrology*, *13*(1), e2157. https://doi.org/10.1002/eco.2157

Sinks, I. A., Borde, A. B., Diefenderfer, H. L., & Karnezis, J. P. (2021). Assessment of Methods to Control Invasive Reed Canarygrass (Phalaris arundinacea) in Tidal Freshwater Wetlands. *Natural Areas Journal*, *41*(3), 172–185. https://doi.org/10.3375/043.041.0303

Stewart, D., Hennigar, D., Ingham, R., & Balke, E. (2022). Factors influencing the persistence of created tidal marshes in the Fraser River Estuary. Ducks Unlimited Canada, Surrey, British Columbia, Canada

Stewart, D., Hood, W. G., & Martin, T. G. (2023). Undetected but Widespread: The Cryptic Invasion of Non-Native Cattail (*Typha*) in a Pacific Northwest Estuary. Estuaries and Coasts, 1-16. https://doi.org/10.1007/s12237-023-01171-4

Stoddard, J. L., Larsen, D. P., Hawkins, C. P., Johnson, R. K., & Norris, R. H. (2006). Setting Expectations for the Ecological Condition of Streams: The Concept of Reference Condition. *Ecological Applications*, *16*(4), 1267–1276. https://doi.org/10.1890/1051-0761(2006)016[1267:SEFTEC]2.0.CO;2

Tilman, D. (1999). The ecological consequences of changes in biodiversity: A search for general principles. *Ecology*, *80*(5), 1455–1474.

Tilman, D., Reich, P. B., & Knops, J. M. H. (2006). Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature*, *441*(7093), 629–632. <https://doi.org/10.1038/nature04742>

Townsend, L., & Hebda, R. J. (2013). Pollen and macro-fossil assemblages in disturbed urban wetlands on South Vancouver Island reveal recent invasion of reed Canarygrass (Phalaris arundinacea) and guide restoration. Restoration Ecology, 21(1), 114-123.

Turner, N. (2014). *Ancient Pathways, Ancestral Knowledge: Ethnobotany and Ecological Wisdom of Indigenous Peoples of Northwestern North America*. McGill-Queen’s Press - MQUP.

Underwood, A. J., Chapman, M. G., & Connell, S. D. (2000). Observations in ecology: You can’t make progress on processes without understanding the patterns. *Journal of Experimental Marine Biology and Ecology*, *250*(1), 97–115. https://doi.org/10.1016/S0022-0981(00)00181-7

Waller, L. P., Allen, W. J., Barratt, B. I. P., Condron, L. M., França, F. M., Hunt, J. E., Koele, N., Orwin, K. H., Steel, G. S., Tylianakis, J. M., Wakelin, S. A., & Dickie, I. A. (2020). Biotic interactions drive ecosystem responses to exotic plant invaders. *Science*, *368*(6494), 967–972. https://doi.org/10.1126/science.aba2225

Whittaker, R. H. (1975). *Communities and Ecosystems* (2nd ed.). Macmillan.

# Supplemental

**Table S1** A total of 27 plots sampled in 1979 and 1999 were not sampled in 2019, mostly due to issues of accessibility. Transect names and plot ID of plots omitted follow Fig. 3 in Bradfield & Porter (1982)

|  |  |  |
| --- | --- | --- |
| **Transect** | **1979/1999**  **Plot No.** | **Reason omitted in 2019** |
| Q | 1-7 | Transect in dense riparian thicket overgrown with Himalayan blackberry |
| R | 8 | Plot on lower bench (> 1 m lower than marsh platform), vegetation no longer exists |
| R | 17-19 | Plots in 1979 & 1999 sampled across a channel. Ended transect in 2019 at channel edge. |
| S | 33-36 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |
| T | 45 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |
| U | 51-52 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |
| V | 53 | Plot 53 only plot across a channel. Increased channel width and likely erosion made crossing this channel dangerous; omitted plot in 2019. |
| V | 54, 70-71 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |
| W | 89-92 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |
| X | 93 | Transect length in 2019 was shorter than in 1979/1999. Suspect combination of erosion and offset transect relocation altered sampling distance. |

**Table S2** Species indicator analysis of cluster groups using Bray-Curtis distance identifies the same dominant species in each assemblage type (Sedge, Fescue, Bogbean), however Bray-Curtis distance identifies different associated indicator species than those identified by Euclidean distance (Table 1)

|  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  |  |  |  |  |  |  |  |  |
|  | **1979** | |  | **1999** | |  | **2019** | |
| **Cluster Group Name** | **Species** | **p-value** |  | **Species** | **p-value** |  | **Species** | **p-value** |
|  |  |  |  |  |  |  |  |  |
| "Sedge" | *Carex lyngbyei* | < 0.01 |  | *Carex lyngbyei* | < 0.01 |  | *Carex lyngbyei* | < 0.01 |
| *Sagittaria latifolia* | < 0.01 |  | *Agrostis stolonifera* | < 0.01 |  | *Mentha canadensis* | 0.03 |
| *Schoenoplectus tabernaemontani* | < 0.01 |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |
| "Fescue" | *Schedonorus arundinaceus* | < 0.01 |  | *Schedonorus arundinaceus* | < 0.01 |  | *Phalaris arundinacea* | < 0.01 |
| *Salix lucida* | < 0.01 |  | *Phalaris arundinacea* | 0.02 |  | *Schedonorus arundinaceus* | < 0.01 |
| *Lathyrus palustris* | < 0.01 |  |  |  |  |  |  |
| *Equisetum palustre* | < 0.01 |  |  |  |  |  |  |
| *Impatiens capensis* | < 0.01 |  |  |  |  |  |  |
| *Sidalcea hendersonii* | < 0.01 |  |  |  |  |  |  |
| *Platanthera dilatata* | 0.02 |  |  |  |  |  |  |
|  |  |  |  |  |  |  |  |  |
| "Bogbean" | *Menyanthes trifoliata* | < 0.01 |  | *Menyanthes trifoliata* | < 0.01 |  | *Mentha aquatica* | < 0.01 |
| *Myosotis scorpioides* | < 0.01 |  | *Leersia oryzoides* | < 0.01 |  | *Menyanthes trifoliata* | < 0.01 |
| *Juncus articulatus* | < 0.01 |  | *Mentha aquatica* | < 0.01 |  | *Lysimachia thyrsiflora* | < 0.01 |
| *Lythrum salicaria* | < 0.01 |  | *Bidens cernua* | < 0.01 |  | *Salix lucida* | < 0.01 |
| *Lysimachia thyrsiflora* | < 0.01 |  | *Lysimachia thyrsiflora* | < 0.01 |  | *Eleocharis palustris* | < 0.01 |
| *Trifolium wormskioldii* | < 0.01 |  | *Juncus articulatus* | < 0.01 |  | *Juncus articulatus* | < 0.01 |
| *Lilaeopsis occidentalis* | < 0.01 |  | *Juncus oxymeris* | 0.02 |  | *Galium trifidum* | 0.01 |
| *Mentha aquatica* | 0.01 |  | *Myosotis scorpioides* | 0.02 |  | *Bidens cernua* | 0.01 |
|  |  |  | Poaceae (unidentified sp.) | 0.01 |  |  |  |
|  |  |  | *Deschampsia caespitosa* | 0.01 |  |  |  |
|  |  |  | *Sagittaria latifolia* | 0.05 |  |  |  |

**Table S3** Bootstrapping 18 randomly selected plots 10 times shows consistent overall trend in loss of species and alpha diversity over time, and overall increase in beta diversity between 1979 and 2019 in all assemblages and across the entire Ladner Marsh plant community. Therefore, loss of plots due to sampling re-location or how number of plots clustered into assemblages is not expected to affect loss of species or plot-based diversity metrics

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
|  | **Plot-level components** | |  | **Diversity components** | | |
| **Assemblage** | **No. quadrats** | **No. species** |  | **α diversity** | **α diversity sd** | **β diversity** |
| **Sedge** |  |  |  |  |  |  |
| 1979 | 18 | 32.3 |  | 10.67 | 2.34 | 3.03 |
| 1999 | 18 | 31.6 |  | 8.31 | 1.98 | 3.81 |
| 2019 | 18 | 30.8 |  | 8.18 | 2.51 | 3.77 |
|  |  |  |  |  |  |  |
| **Fescue** |  |  |  |  |  |  |
| 1979 | 18 | 43 |  | 13.0 | 3.9 | 3.3 |
| 1999 | 18 | 36 |  | 9.7 | 3.9 | 3.8 |
| 2019 | 18 | 27 |  | 5.8 | 2.8 | 4.6 |
|  |  |  |  |  |  |  |
| **Bogbean** |  |  |  |  |  |  |
| 1979 | 18 | 32 |  | 12.8 | 3.6 | 2.5 |
| 1999 | 18 | 36 |  | 11.5 | 2.9 | 3.1 |
| 2019 | 18 | 31 |  | 10.5 | 1.9 | 3.0 |
|  |  |  |  |  |  |  |
| **Total** |  |  |  |  |  |  |
| 1979 | 54 | 48 |  | 12.2 | 3.5 | 3.9 |
| 1999 | 54 | 42 |  | 10.0 | 3.4 | 4.2 |
| 2019 | 54 | 42 |  | 8.2 | 3.1 | 5.1 |



**Table S4** Total turnover and rates of species disappearance (loss) was always greater between 1999 and 2019 than between 1979 and 1999. However, fewer species were gained in the Bogbean assemblage 1999-2019 than 1979-1999

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Assemblage** | **Year** | **Total turnover** | **Species Appearance** | **Species Disappearance** |
| Bogbean | 1979-1999 | 0.56 | 0.35 | 0.22 |
| 1999-2019 | 0.60 | 0.28 | 0.32 |
| Fescue | 1979-1999 | 0.46 | 0.20 | 0.27 |
| 1999-2019 | 0.64 | 0.18 | 0.46 |
| Sedge | 1979-1999 | 0.46 | 0.24 | 0.22 |
| 1999-2019 | 0.56 | 0.27 | 0.29 |

**Table S5** Mean cover class values for non-native and native species observed in each assemblage for each sampling period. Overall change from 1979 to 2019 indicates cover abundance decreases (-), cover abundance increases (+), and species gained or lost. For each year, blank spaces indicate no data for the species in that sampling year; blank spaces in ‘Overall Change’ indicate species found only in 1999

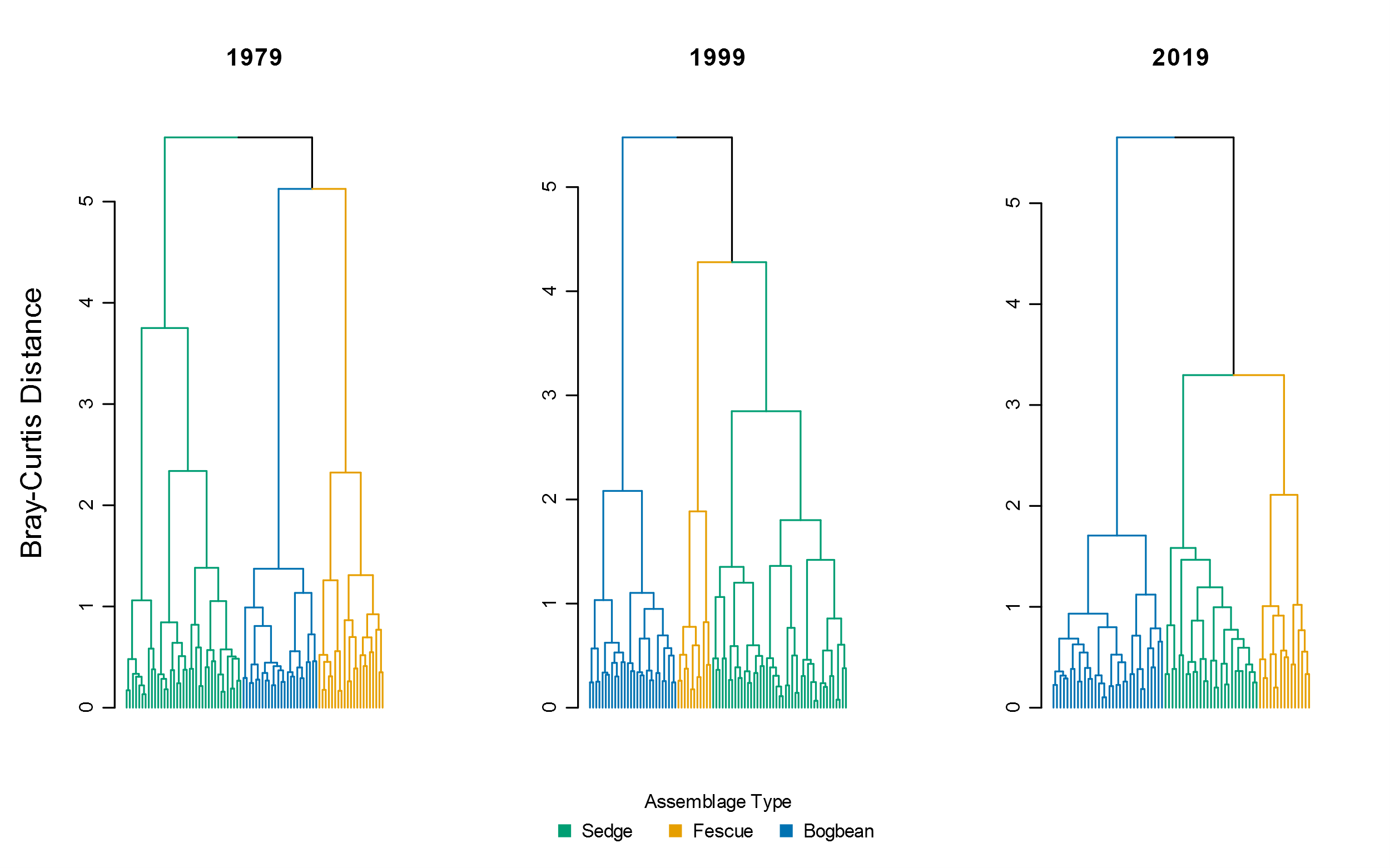
|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Assemblage** | **Status** | **Species** | **1979** | **1999** | **2019** | **Overall Change (1979-2019)** |
| Bogbean | Non-native | *Alisma plantago aquatica* | 0.2 | 0.1 |  | lost |
| *Myosotis scorpioides* | 0.7 | 0.2 | 0.2 | - |
| *Agrostis stolonifera* | 3.2 | 1.5 | 1.3 | - |
| *Lythrum salicaria* | 1.1 | 1.2 | 0.6 | - |
| *Rumex conglomeratus* | 0.1 |  | < 0.1 | - |
| *Mentha aquatica* | 0.4 | 2.3 | 1.8 | + |
| *Iris pseudacorus* |  | 0.3 | 0.2 | gained |
| *Lycopus europaeus* |  |  | < 0.1 | gained |
| *Phalaris arundinacea* |  | 0.1 | < 0.1 | gained |
| *Schedonorus arundinaceus* |  | 0.2 |  |  |
| Native | *Alopecurus geniculatus* | 0.1 |  |  | lost |
| *Deschampsia caespitosa* | 0.3 | 0.2 |  | lost |
| *Equisetum fluviatile* | 1.4 | 1.2 |  | lost |
| *Leersia oryzoides* | 0.3 | 0.3 |  | lost |
| *Lilaeopsis occidentalis* | 0.2 |  |  | lost |
| *Oenanthe sarmentosa* | 0.6 | 0.1 |  | lost |
| *Poa trivialis* | 0.1 |  |  | lost |
| *Sium suave* | 0.6 | 0.2 |  | lost |
| *Caltha palustris* | 0.9 | 0.2 | 0.1 | - |
| *Bidens cernua* | 0.8 | 0.2 | 0.1 | - |
| *Trifolium wormskioldii* | 0.9 | 0.1 | 0.2 | - |
| *Schoenoplectus tabernaemontani* | 0.2 |  | 0.1 | - |
| *Eleocharis palustris* | 0.6 | 0.8 | 0.4 | - |
| *Symphyotrichum subspicatum* | 0.5 | 0.3 | 0.3 | - |
| *Juncus oxymeris* | 0.1 | 0.1 | < 0.1 | - |
| *Mentha canadensis* | 0.5 |  | < 0.1 | - |
| *Platanthera dilatata* | 0.1 | 0.1 | < 0.1 | - |
| *Menyanthes trifoliata* | 3.8 | 3.1 | 3.0 | - |
| *Lysimachia thyrsiflora* | 0.5 | 0.2 | 0.6 | + |
| *Juncus articulatus* | 0.3 | 0.4 | 0.3 | + |
| *Sidalcea hendersonii* | 0.1 |  | 0.1 | + |
| *Carex lyngbyei* | 0.5 | 0.3 | 1.0 | + |
| *Rumex occidentalis* | 0.1 | 0.1 | 0.1 | + |
| *Potentilla anserina-pacifica* | 0.3 | 1.0 | 1.1 | + |
| *Equisetum arvense* |  |  | 0.6 | gained |
| *Galium trifidum* |  |  | 0.4 | gained |
| *Hypericum scouleri* |  |  | < 0.1 | gained |
| *Impatiens capensis* |  | 0.4 | 0.3 | gained |
| *Juncus acuminatus* |  |  | < 0.1 | gained |
| *Lathyrus palustris* |  | 0.1 | 0.5 | gained |
| *Lysichiton americanum* |  |  | 0.1 | gained |
| *Salix lasiandra* |  | 0.6 | 0.5 | gained |
| *Salix scouleriana* |  |  | < 0.1 | gained |
| *Typha latifolia* |  | 0.3 | 0.3 | gained |
| *Equisetum palustre* |  | 0.1 |  |  |
| *Equisetum variegatum* |  | 0.1 |  |  |
| *Galium sp.* |  | 0.1 |  |  |
| *Poa palustris* |  | 0.5 |  |  |
| *Poaceae sp.* |  | 0.3 |  |  |
| *Sagittaria latifolia* |  | 0.2 |  |  |
|  |  |  |  |  |  |  |
| **Assemblage** | **Status** | **Species** | **1979** | **1999** | **2019** | **Overall Change (1979-2019)** |
| Fescue | Unknown | *Festuca sp.* | < 0.1 |  |  | lost |
| Non-native | *Alisma plantago aquatica* | 0.1 | 0.2 |  | lost |
| *Mentha aquatica* | 0.3 | 0.1 |  | lost |
| *Myosotis scorpioides* | 0.3 | < 0.1 |  | lost |
| *Schedonorus arundinaceus* | 1.6 | 0.9 | 0.7 | - |
| *Lythrum salicaria* | 0.4 | 0.6 | 0.4 | + |
| *Agrostis stolonifera* | 0.3 | 0.8 | 0.6 | + |
| *Phalaris arundinacea* | 0.1 | 0.2 | 1.1 | + |
| *Cirsium arvense* |  | < 0.1 | 0.1 | gained |
| *Iris pseudacorus* |  | 0.2 | 0.2 | gained |
| *Lycopus europaeus* |  |  | 0.1 | gained |
| Native | *Alopecurus geniculatus* | < 0.1 |  |  | lost |
| *Bidens cernua* | 0.2 | 0.5 |  | lost |
| *Deschampsia caespitosa* | 0.6 | 0.1 |  | lost |
| *Dulichium arundinaceum* | 0.1 |  |  | lost |
| *Eleocharis palustris* | 1.0 | 0.3 |  | lost |
| *Equisetum palustre* | 0.8 | 0.1 |  | lost |
| *Galium trifidum* | < 0.1 |  |  | lost |
| *Hypericum formosum* | 0.1 |  |  | lost |
| *Juncus articulatus* | 0.5 | 0.1 |  | lost |
| *Leersia oryzoides* | 0.1 | 0.2 |  | lost |
| *Lilaeopsis occidentalis* | 0.2 |  |  | lost |
| *Erythranthe scouleri* | < 0.1 |  |  | lost |
| *Oenanthe sarmentosa* | 0.2 | 0.3 |  | lost |
| *Platanthera dilatata* | 0.2 | 0.0 |  | lost |
| *Poa palustris* | 0.6 | 1.7 |  | lost |
| *Poa trivialis* | 0.3 |  |  | lost |
| *Polygonum hydropiper* | < 0.1 |  |  | lost |
| *Sagittaria latifolia* | < 0.1 | 0.2 |  | lost |
| *Salix sp.* | < 0.1 |  |  | lost |
| *Sium suave* | 0.1 | 0.2 |  | lost |
| *Symphyotrichum subspicatum* | 0.6 | 0.2 |  | lost |
| *Trifolium wormskioldii* | 0.7 | 0.5 |  | lost |
| *Menyanthes trifoliata* | 1.9 | 1.3 | 0.1 | - |
| *Caltha palustris* | 0.7 | 0.4 | 0.1 | - |
| *Salix lasiandra* | 1.0 | 0.4 | 0.1 | - |
| *Carex lyngbyei* | 0.8 | 1.4 | 0.1 | - |
| *Potentilla anserina-pacifica* | 0.5 | 0.6 | 0.2 | - |
| *Sidalcea hendersonii* | 0.4 | 0.2 | 0.2 | - |
| *Mentha canadensis* | 0.2 | 0.2 | 0.1 | - |
| *Typha latifolia* | 0.7 | 0.4 | 0.4 | - |
| *Hordeum brachyantherum* | 0.2 |  | 0.1 | - |
| *Equisetum fluviatile* | 0.6 | 0.4 | 0.4 | - |
| *Lathyrus palustris* | 0.6 | 0.2 | 0.6 | + |
| *Rumex occidentalis* | 0.1 | 0.2 | 0.1 | + |
| *Impatiens capensis* | 0.3 | 0.4 | 0.6 | + |
| *Equisetum arvense* |  |  | 0.4 | gained |
| *Juncus effusus* |  |  | 0.1 | gained |
| *Lysichiton americanum* |  |  | 0.1 | gained |
| *Myrica gale* |  |  | 0.2 | gained |
| *Salix scouleriana* |  |  | 0.2 | gained |
| *Lysimachia thyrsiflora* | 0.1 | 0.3 | 0.1 |  |
| *Schoenoplectus tabernaemontani* | 0.1 | 0.2 | 0.1 |  |
| *Asteraceae sp.* |  | < 0.1 |  |  |
| *Carex sp.* |  | 0.1 |  |  |
| *Galium sp.* |  | < 0.1 |  |  |
| *Juncus oxymeris* |  | 0.1 |  |  |
| *Salix sitchensis* |  | < 0.1 |  |  |
|  |  |  |  |  |  |  |
| **Assemblage** | **Status** | **Species** | **1979** | **1999** | **2019** | **Overall Change (1979-2019)** |
| Sedge | Unknown | *Galium sp.* |  | < 0.1 |  |  |
| Non-native | *Alisma plantago aquatica* | 0.4 | 0.1 |  | lost |
| *Myosotis scorpioides* | < 0.1 |  |  | lost |
| *Agrostis stolonifera* | 1.9 | 2.3 | 1.3 | - |
| *Lythrum salicaria* | 0.3 | 0.3 | 0.4 | + |
| *Schedonorus arundinaceus* | 0.1 | 0.1 | 0.2 | + |
| *Iris pseudacorus* |  | 0.1 | 0.3 | gained |
| *Lycopus europaeus* |  | < 0.1 | 0.1 | gained |
| *Mentha aquatica* |  | 0.2 | 0.5 | gained |
| *Phalaris arundinacea* |  |  | 0.1 | gained |
| *Cirsium arvense* |  | < 0.1 |  |  |
| Native | *Deschampsia caespitosa* | 0.2 |  |  | lost |
| *Leersia oryzoides* | 0.2 | 0.2 |  | lost |
| *Lilaeopsis occidentalis* | 0.1 | 0.1 |  | lost |
| *Erythranthe scouleri* | 0.1 |  |  | lost |
| *Oenanthe sarmentosa* | 0.7 | 0.4 |  | lost |
| *Platanthera dilatata* | 0.1 | < 0.1 |  | lost |
| *Poa palustris* | 1.0 | 0.2 |  | lost |
| *Puccinellia pauciflora* | < 0.1 |  |  | lost |
| *Sium suave* | 0.6 | 0.2 |  | lost |
| *Caltha palustris* | 1.1 | 0.5 | < 0.1 | - |
| *Equisetum fluviatile* | 0.9 | 0.6 | < 0.1 | - |
| *Mentha canadensis* | 0.3 | 0.2 | < 0.1 | - |
| *Schoenoplectus tabernaemontani* | 0.7 | 0.1 | 0.1 | - |
| *Trifolium wormskioldii* | 0.4 | 0.1 | 0.1 | - |
| *Sagittaria latifolia* | 0.4 | 0.1 | 0.1 | - |
| *Bidens cernua* | 0.5 | 0.1 | 0.2 | - |
| *Eleocharis palustris* | 0.8 | 0.4 | 0.4 | - |
| *Menyanthes trifoliata* | 0.3 | 0.7 | 0.2 | - |
| *Carex lyngbyei* | 3.0 | 3.0 | 1.9 | - |
| *Typha latifolia* | 0.6 | 0.4 | 0.4 | - |
| *Symphyotrichum subspicatum* | 0.3 | 0.1 | 0.3 | - |
| *Rumex occidentalis* | 0.1 | 0.2 | 0.1 | - |
| *Lysimachia thyrsiflora* | 0.1 |  | 0.1 | + |
| *Sidalcea hendersonii* | 0.1 | 0.1 | 0.2 | + |
| *Potentilla anserina-pacifica* | 0.3 | 0.7 | 0.8 | + |
| *Rumex conglomeratus* | < 0.1 |  | 0.1 | + |
| *Lathyrus palustris* | 0.1 | 0.3 | 0.5 | + |
| *Impatiens capensis* | 0.1 | 1.1 | 0.9 | + |
| *Salix lasiandra* | < 0.1 | < 0.1 | 0.3 | + |
| *Equisetum arvense* |  |  | 0.7 | gained |
| *Galium palustre* |  |  | < 0.1 | gained |
| *Galium trifidum* |  |  | 0.1 | gained |
| *Hypericum scouleri* |  |  | 0.1 | gained |
| *Juncus articulatus* |  |  | < 0.1 | gained |
| *Juncus oxymeris* |  |  | < 0.1 | gained |
| *Scirpus microcarpus* |  |  | 0.1 | gained |
| *Equisetum palustre* |  | 0.2 |  |  |
| *Lysichiton americanum* |  | < 0.1 |  |  |
| *Salix sitchensis* |  | 0.1 |  |  |

**Table S6** Mean cover class values for all species recorded in each observation dataset, averaged across all plots. Across all observations, there was a net loss of five native species, and a net gain of two non-native species. Plants identified only to Family or genus are included for reference, but were not considered as a part of species net gain/loss.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Species** | **1979** | **1999** | **2019** | **Status** | **Change** |
| *Alisma plantago aquatica* | 0.22 | 0.12 | 0.00 | Non-native | lost |
| *Myosotis scorpioides* | 0.28 | 0.06 | 0.08 | - |
| *Schedonorus arundinaceus* | 0.59 | 0.44 | 0.24 | - |
| *Agrostis stolonifera* | 1.63 | 1.54 | 1.11 | - |
| *Lythrum salicaria* | 0.50 | 0.59 | 0.49 | - |
| *Rumex conglomeratus* | 0.02 | 0.00 | 0.05 | + |
| *Mentha aquatica* | 0.20 | 0.60 | 0.88 | + |
| *Phalaris arundinacea* | 0.02 | 0.07 | 0.30 | + |
| *Cirsium arvense* | 0.00 | 0.02 | 0.01 | gained |
| *Iris pseudacorus* | 0.00 | 0.18 | 0.23 | gained |
| *Lycopus europaeus* | 0.00 | 0.00 | 0.07 | gained |
|  |  |  |  |  |  |
| *Alopecurus geniculatus* | 0.02 | 0.00 | 0.00 | Native | lost |
| *Deschampsia caespitosa* | 0.37 | 0.09 | 0.00 | lost |
| *Dulichium arundinaceum* | 0.02 | 0.00 | 0.00 | lost |
| *Equisetum palustre* | 0.27 | 0.13 | 0.00 | lost |
| *Leersia oryzoides* | 0.18 | 0.24 | 0.00 | lost |
| *Lilaeopsis occidentalis* | 0.13 | 0.04 | 0.00 | lost |
| *Erythranthe scouleri* | 0.05 | 0.00 | 0.00 | lost |
| *Oenanthe sarmentosa* | 0.50 | 0.29 | 0.00 | lost |
| *Poa palustris* | 0.61 | 0.89 | 0.00 | lost |
| *Poa trivialis* | 0.13 | 0.00 | 0.00 | lost |
| *Polygonum hydropiper* | 0.01 | 0.00 | 0.00 | lost |
| *Puccinellia pauciflora* | 0.01 | 0.00 | 0.00 | lost |
| *Sium suave* | 0.44 | 0.17 | 0.00 | lost |
| *Caltha palustris* | 0.90 | 0.39 | 0.05 | - |
| *Platanthera dilatata* | 0.12 | 0.04 | 0.01 | - |
| *Equisetum fluviatile* | 0.90 | 0.62 | 0.12 | - |
| *Trifolium wormskioldii* | 0.63 | 0.29 | 0.09 | - |
| *Mentha canadensis* | 0.29 | 0.16 | 0.05 | - |
| *Sagittaria latifolia* | 0.18 | 0.13 | 0.04 | - |
| *Schoenoplectus tabernaemontani* | 0.35 | 0.10 | 0.08 | - |
| *Bidens cernua* | 0.46 | 0.29 | 0.14 | - |
| *Eleocharis palustris* | 0.82 | 0.44 | 0.30 | - |
| *Hordeum brachyantherum* | 0.06 | 0.00 | 0.03 | - |
| *Symphyotrichum subspicatum* | 0.44 | 0.22 | 0.22 | - |
| *Juncus articulatus* | 0.24 | 0.11 | 0.12 | - |
| *Carex lyngbyei* | 1.61 | 1.79 | 1.14 | - |
| *Menyanthes trifoliata* | 1.68 | 1.46 | 1.22 | - |
| *Typha latifolia* | 0.49 | 0.34 | 0.36 | - |
| *Sidalcea hendersonii* | 0.20 | 0.11 | 0.16 | - |
| *Salix lasiandra* | 0.37 | 0.30 | 0.32 | - |
| *Lysimachia thyrsiflora* | 0.20 | 0.18 | 0.27 | + |
| *Rumex occidentalis* | 0.09 | 0.15 | 0.12 | + |
| *Potentilla anserina-pacifica* | 0.35 | 0.76 | 0.76 | + |
| *Lathyrus palustris* | 0.23 | 0.20 | 0.50 | + |
| *Juncus oxymeris* | 0.01 | 0.06 | 0.03 | + |
| *Impatiens capensis* | 0.16 | 0.67 | 0.59 | + |
| *Galium trifidum* | 0.01 | 0.00 | 0.18 | + |
| *Equisetum arvense* | 0.00 | 0.00 | 0.59 | gained |
| *Galium palustre* | 0.00 | 0.00 | 0.01 | gained |
| *Juncus acuminatus* | 0.00 | 0.00 | 0.01 | gained |
| *Juncus effusus* | 0.00 | 0.00 | 0.01 | gained |
| *Lysichiton americanum* | 0.00 | 0.01 | 0.05 | gained |
| *Myrica gale* | 0.00 | 0.00 | 0.05 | gained |
| *Salix scouleriana* | 0.00 | 0.00 | 0.05 | gained |
| *Scirpus microcarpus* | 0.00 | 0.00 | 0.03 | gained |
| *Equisetum variegatum* | 0.00 | 0.02 | 0.00 | NA |
| *Hypericum scouleri* | 0.04 | 0.00 | 0.04 | NA |
| *Salix sitchensis* | 0.00 | 0.04 | 0.00 | NA |
|  |  |  |  |  |  |
| *Festuca* sp. | 0.01 | 0.00 | 0.00 | Unknown (Not identified to species) |  |
| *Salix* sp. | 0.01 | 0.00 | 0.00 |  |
| *Asteraceae* sp. | 0.00 | 0.01 | 0.00 |  |
| *Carex* sp. | 0.00 | 0.02 | 0.00 |  |
| *Galium* sp. | 0.00 | 0.04 | 0.00 |  |
| *Poaceae* sp. | 0.00 | 0.06 | 0.00 |  |

**Table S7** All species recorded in 1979, 1999, and 2019, their synonymous nomenclature, and endemic status according to the Electronic Atlas of the Flora of British Columbia (E-Flora BC, https://ibis.geog.ubc.ca/biodiversity/eflora/)

|  |  |  |
| --- | --- | --- |
| **Species reported 1979-2019** | **Synonym recorded in 1979 and/or 1999** | **Endemism Status** |
| *Agrostis stolonifera* | *Agrostis alba* | Non-native |
| *Alisma plantago-aquatica* |  | Non-native |
| *Alopecurus geniculatus* |  | Non-native |
| *Bidens cernua* |  | Native |
| *Caltha palustris* |  | Native |
| *Carex lyngbyei* |  | Native |
| *Carex* sp1 |  | NA |
| *Carex* sp2 |  | NA |
| *Cirsium arvense* |  | Non-native |
| *Composite* (unidentified) |  | NA |
| *Deschampsia caespitosa* |  | Native |
| *Dulichium arundinaceum* |  | Native |
| *Eleocharis palustris* |  | Native |
| *Erythranthe scouleri* | *Mimulus guttatus* | Native |
| *Equisetum arvense* |  | Native |
| *Equisetum fluviatile* |  | Native |
| *Equisetum variegatum* |  | Native |
| *Festuca* sp |  | NA |
| *Galium palustre* |  | Native |
| *Galium* sp |  | NA |
| *Galium trifidum* | *Galium cymosum* | Native |
| Grass(unidentified) |  | NA |
| *Hordeum brachyantherum* |  | Native |
| *Hypericum scouleri* | *Hypericum formosum* | Native |
| *Impatiens capensis* |  | Non-native |
| *Iris pseudacorus* |  | Non-native |
| *Juncus acuminatus* |  | Native |
| *Juncus articulatus* |  | Native |
| *Juncus effusus* |  | Native |
| *Juncus oxymeris* |  | Native |
| *Lathyrus palustris* |  | Native |
| *Leersia oryzoides* |  | Native |
| *Lilaea scilloides* |  | Native |
| *Lilaeopsis occidentalis* |  | Native |
| *Lycopus europaeus* |  | Non-native |
| *Lysichiton americanus* |  | Native |
| *Lysimachia thyrsiflora* |  | Native |
| *Lythrum salicaria* |  | Non-native |
| *Mentha aquatica* | *Mentha citrata* | Non-native |
| *Mentha canadensis* |  | Non-native |
| *Menyanthes trifoliata* |  | Native |
| *Myosotis scorpioides* |  | Non-native |
| *Myrica gale* |  | Native |
| *Oenanthe sarmentosa* |  | Native |
| *Phalaris arundinacea* |  | Non-native |
| *Platanthera dilatata* |  | Native |
| *Poa palustris* |  | Native |
| *Poa trivialis* |  | Non-native |
| *Polygonum hydropiper* |  | Non-native |
| *Potentilla pacifica* |  | Native |
| *Puccinellia pauciflora* |  | Native |
| *Rumex conglomeratus* |  | Non-native |
| *Rumex occidentalis* |  | Native |
| *Sagittaria latifolia* |  | Native |
| *Salix lasiandra* |  | Native |
| *Salix scouleriana* |  | Native |
| *Salix sitchensis* |  | Native |
| *Salix* sp |  | NA |
| *Schedonorus arundinaceus* | *Festuca arundinacea* | Non-native |
| *Schoenoplectus tabernaemontani* | *Scirpus validus* | Native |
| *Scirpus microcarpus* |  | Native |
| *Sidalcea hendersonii* |  | Native |
| *Sium suave* |  | Native |
| *Sonchus arvensis* |  | Non-native |
| *Symphyotrichum subspicatum* | *Aster eatonii* | Native |
| *Trifolium wormskioldii* |  | Native |
| *Typha latifolia* |  | Native |
| *Zannichellia palustris* |  | Native |

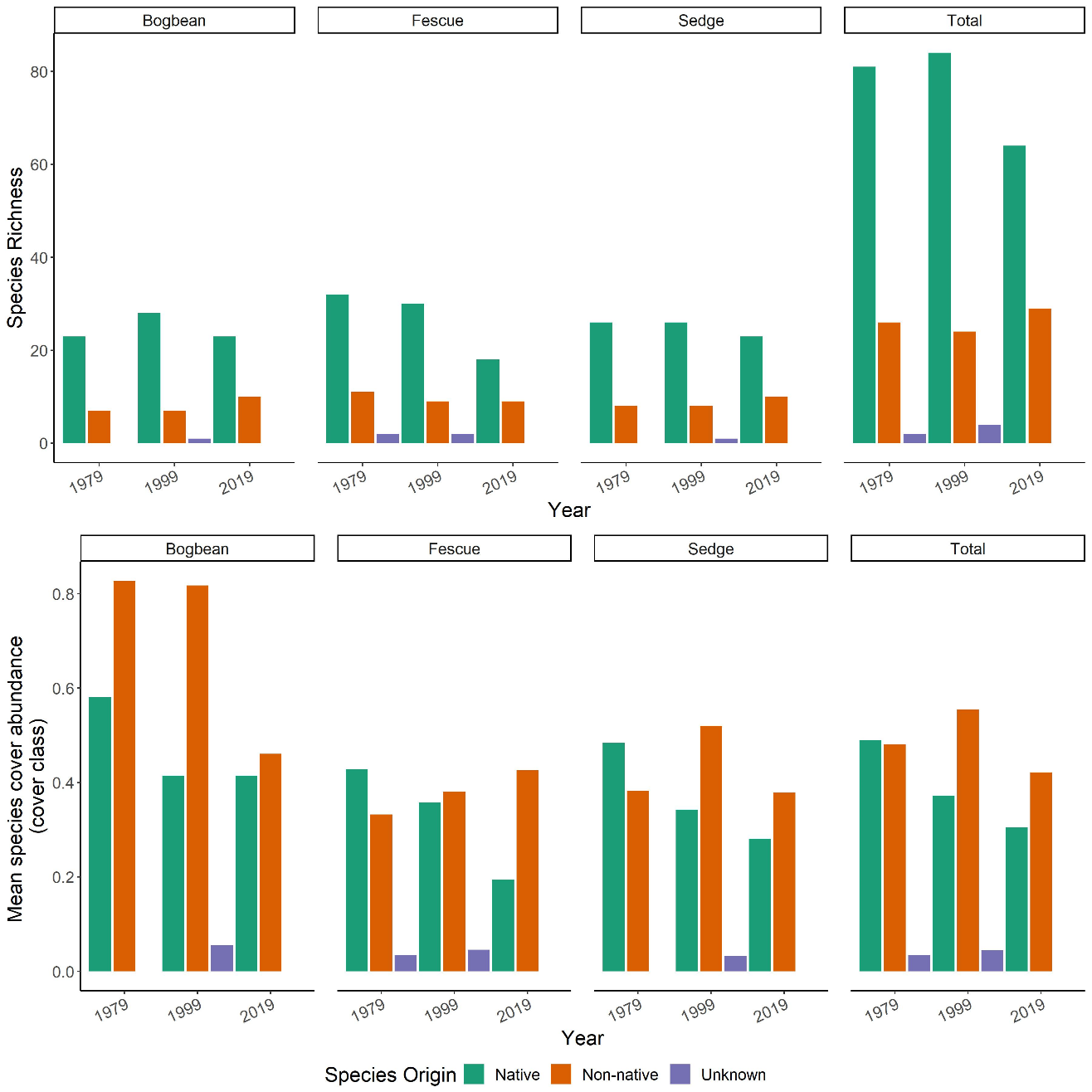


**Fig. S1** Cluster analysis using Bray-Curtis distance measure shows similar trends of increasing homogeneity within assemblages as when using Euclidean distance (Fig. 3)

A diagram of a graph

Description automatically generated with medium confidence

**Fig. S2**. Percentage of plots clustered in each assemblage calculated for each transect. Relatively even percentages of plots within each assemblage along a single transect support accuracy of transect relocation and/or plant community stability (e.g., transects W,X). Discrepancies (e.g., transects U, V) may be indicative of spatial inaccuracies in transect relocation and/or greater turnover within a given sampling year.



**Fig. S3** Top panel: Loss of native species richness over time across all assemblages is largely driven by loss of native species from the Fescue Assemblage. However, native species richness does not change substantially in the other two assemblages. Bottom panel: Native species cover is decreasing on average across all assemblages. Non-native species cover shows a more variable pattern of change, although the ratio of native to non-native cover in Bogbean assemblage becomes more even by 2019. ‘Unknown’ species origin represents species identified only to genus, and assessment of native status cannot be made.